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Forest cover impact on water related ecosystem services

Methods and application at the regional scale (Wallonia, Belgium)

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FOREST COVER IMPACT ON WATER RELATED ECOSYSTEM SERVICES

Methods and application at the regional scale (Wallonia, Belgium)

Thesis submitted in fulfillment of the requirements for the degree of Doctor of Sciences (PhD) by

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Abstract

The concept of Ecosystem Services (ES) highlights Humanity's dependence on ecosystems for its survival and well-being in a global context of ecosystems' degradation. One model that has been widely used represents ES at the centre of a 'cascade' (Haines-Young and Potschin, 2010a), flowing from the ecosystems biophysical structures and processes to human well-being. Among research needs regarding ES, there is a crucial one for accurately quantifying every component of the ES 'cascade' through suitable indicators. While current policy-driven initiatives of ES assessments and mapping are often based on methods relying on simple land cover proxies, research is needed to propose indicators that can easily be mapped, but better reflect the underlying complexity of processes underpinning ES supply.

Among ES, those related to water are of prime importance. Literature regarding forest cover effect on water related processes is relatively abundant. However, the combined effect of these processes on hydrological ES is less evident given the ecosystems' complexity and heterogeneity at the landscape scale. Questions related to the integrated effect of mixed land uses and land covers at the landscape scale and regarding the forests' position in the landscape (i.e. within riparian zone or within the whole catchment) where its effect on hydrological ES is the strongest remain unanswered. Finally, global changes push for renewing the studies of the ecosystems' effect on hydrological ES.

The main objective addressed in this research is to assess the impact of forest cover on hydrological ES in Wallonia (Belgium). In particular, the effect of forest cover on instream water supply and flood protection is studied in terms of quantity, quality and timing. Along with this thematic objective, transversal methodological objectives are pursued: to ensure replicability of the methods and to broaden the scope of the results, moving towards land planning oriented results.

Our main results show that forest cover effect on instream water supply in terms of quantity is negative when studying water yield, whereas a significant positive effect of forest cover in low flows is demonstrated. Studying baseflow relationship with forest cover lets us assume that local site conditions (soil

types, topography, forest management) have a major impact on specific volume. Regarding flood protection, forest cover is negatively linked with the flashy behaviour of the catchment thus a positive effect on the flood protection ES. Climatic factors and rainfall in particular are often significantly linked to hydrological indicators and can be considered as main drivers of instream water supply and flood protection.

Regarding instream water supply in terms of quality, one main result is that forest cover is systematically positively correlated with higher water quality whether when describing it through nine physico-chemical variables or through two biological indices (based on diatoms and macroinvertebrates). In both studies, forest cover explains about one third of the variability of water quality (and around 10% when spatial autocorrelation is controlled) at the regional scale. Results also show that unlike needle-leaved forest cover, broad-leaved forest cover presents an independent effect from ecological variables on physico-chemical water quality. Another important insight of this study is that physico-chemical water quality is one of the main drivers of biological water quality, and that anthropogenic pressures often explain a relatively important part of biological water quality. Results on biological water quality show that the proportion of forest cover in each catchment at the regional scale and across every ecoregions except for the Loam region is more positively correlated with high water quality than the proportion of forest cover in the riparian zone only.

Results regarding forest cover effect on studied hydrological ES in terms of quantity and timing make us question the use of LULC based matrix approach to assess and map hydrological ES at a complex landscape scale. However, the strong link between forest cover in catchment and water quality allows being more confident when using simple land cover proxies to map ecosystem services related to water quality.

Working with "real-life" catchments presents the advantage to fit the spatial scale for drawing land-planning recommendations. Results at the regional scale and across ecoregions lead us to recommend riparian forests protection in the Loam region (where they are left) but the overall forest catchment effect on water quality (whether physico-chemical or biological) suggests that catchment-wide impacts and a fortiori catchment-wide protection measures are the main drivers of rivers' ecological water quality.

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X

"It is our collective and individual responsibility to preserve and tend to the environment in which we all live."

Dalai Lama

And because this PhD journey was simultaneous with an inspiring teaching experience,

"Nothing under the sun is greater than education. By educating one person and sending him into the society of his generation, we make a contribution extending a hundred generations to come."

Kano Jigoro, founder of Judo

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AET	Actual Evapotranspiration (m^3/s)
BISE	Biodiversity Information System for Europe
BFI	Baseflow Index
CICES	Common International Classification of Ecosystem Services
CBD	Convention on Biological Diversity
Cl	Chloride (mg/l)
DO	Dissolved Oxygen (mgO2/l)
DOC	Dissolved Organic Carbon (mgC/l)
ES	Ecosystem Services
FI	Flashiness Index
HES	Hydrological Ecosystem Services
IBGN	Standardized Global Biological Index
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPS	Specific Polluosensitivity Index
IUCN	International Union for the Conservation of Nature
LULC	Land Use and Land Cover
MAES	Mapping and Assessment of Ecosystem Services
MLR	Multiple Linear Regression
NGI	Belgian National Geographic Institute
NH4	Ammonium (mgN/l)
NO2	Nitrites (mgN/l)
NO3	Nitrates (mgN/l)
P	Precipitation (m^3/s)
PCA	Principal Components Analysis
Qxs	Specific discharge exceeded x% of the time (m/s)
SO4	Sulphate (mg/l)
TEEB	The Economics of Ecosystems and Biodiversity
TP	Total phosphorus (mgP/l)
RDA	Redundancy Analysis
Vs	Specific Volume (m^3/m^2)
WalES	Walloon Ecosystem services

WCS	Wildlife Conservation Society
EU-WFD	European Water Framework Directive
WPS	Walloon Public Service
WWF	World Wide Fund for Nature

Chapter 1 Introduction

1.1 Preamble

This document discusses the impact of forest cover on water related ecosystem services. The present section describes the state of the art, points to knowledge gaps leading to the definition of our thesis' objectives. More specifically, the ecosystem services (ES) concept and its uses are first characterized from the scientific and socio-political points of view. Then, particular research needs regarding this concept are identified. Water related ES are then presented along with current knowledge on the impact of forest cover on these ES. This allows raising unanswered scientific questions. Following this, detailed thematic and methodological objectives are defined and the scope and main assumptions of this PhD presented.

1.2 Ecosystem services

1.2.1 Humanity's dependency on Nature

While planet boundaries are dramatically being crossed (Loh et al., 2005; Rockström et al., 2009; Steffen et al., 2015) and ecosystems degraded, the Ecosystem Services concept has been promoted as a mean to raise awareness about the importance of preserving ecosystems and biodiversity (Millennium Ecosystem Assessment, 2005a).

This concept of ES, that can be defined as the "benefit people obtain from nature" (Millennium Ecosystem Assessment, 2003), highlights Humanity's dependence on ecosystems and ecosystem processes for its survival and well-being. The idea of human dependency on Nature is not new [see Plato's descriptions on the effects of deforestation on soil erosion and the drying of springs in 400 BC (Daily et al., 1997) or Pliny the Elder observation of the links between deforestation, rainfall, and the occurrence of torrents in the first century AD (Andréassian, 2004a)]. However, these "ecosystem services" started to be explicitly considered in the mid-1960s early 1970's, when scientist began to address the societal value of nature's functions [e.g. King

(1966), Helliwel (1969), Odum and Odum (1972)]. The term of "ecosystem services" was introduced by Ehrlich and Ehrlich (1982) and several important scientific contributions followed [e.g. Daily et al. (1997) or Costanza et al. (1997)]. Gómez-Baggethun et al. (2010) present the historic development of the conceptualization of ES. The ES concept took on an international and political dimension with the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2003, 2005a).

1.2.2 An evolving concept

Ecosystem functions, services, natural capital have been defined several times during the last decades, still, there is not a standardized meaning for these concepts (Boyd and Banzhaf, 2007; Fisher et al., 2009; Haines-Young and Potschin, 2010b; La Notte et al., 2017; Wallace, 2007).

Barnaud et al. (2017, submitted) insist on the necessity to adopt a constructivist perspective. Indeed, the concept of ES is not stabilized and still evolving. ES are social constructions, representing inherently subjective perceptions of human-nature relations [Latour, 2004 cited in Barnaud et al. (2017, submitted)]. This instability is reflected in the multiple conceptual frameworks describing these ES at the interface of Ecosystems and Human well-being (Maes et al., 2013; Millennium Ecosystem Assessment, 2003; Pascual et al., 2017). One of them that has been widely used represents this concept in the form of a 'cascade' model [(Haines-Young and Potschin, 2010a), Figure 1-1]. This framework is based on the idea that a sort of 'production chain' starts from the ecosystems biophysical structures and processes which lead to functions that create services providing benefits and socio-economic values to human beings. Human society retroacts on ecosystems through pressures but also restoration actions.

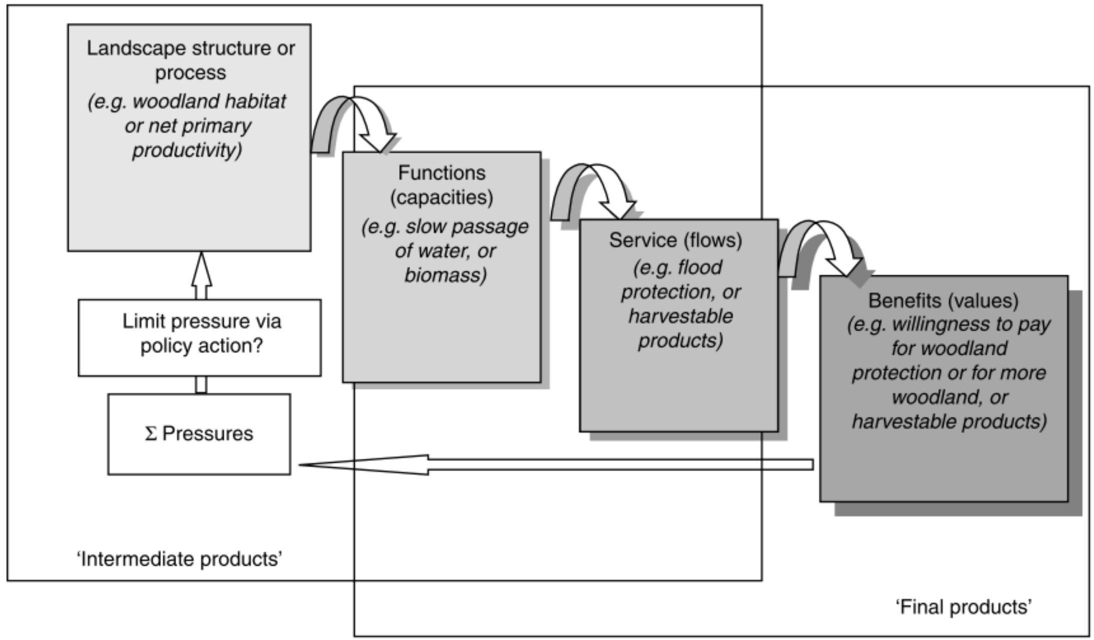


Figure 1-1. The relationship between ecosystem structure and biodiversity, ecosystem function and human (Haines-Young and Potschin, 2010a).

At the basis of the cascade model, ecosystem structure represents “the biophysical architecture of an ecosystem”, or in other words, the “static ecosystem characteristics: spatial and aspatial structure, composition and distribution of biophysical elements” (e.g. land use, standing crop, leaf area, species composition,…). Ecosystem processes represent the dynamic ecosystem characteristics and can be defined as “complex interactions among biotic and abiotic elements causing physical, chemical and biological changes or reactions” (Englund et al., 2017). These processes can be physical (e.g. infiltration of water, sediment movement), chemical (e.g. reduction, oxidation) or biological (e.g. photosynthesis, microbiota decomposition). Even if uncertainties remain on the effect of complexity of biodiversity components on the ecosystem functioning that underpins ES (Balvanera et al., 2014), the importance of ‘biodiversity’ in underpinning ES is acknowledged (Díaz et al., 2006; Haines-Young and Potschin, 2010a; Millennium Ecosystem Assessment, 2005a). Harrison et al. (2014), in a review aiming at analysing the linkage between biodiversity attributes and 11 ES, show that the majority of the relationships are positive except for the freshwater provision ES where some biotic attributes (such as community attributes of area, age, structure) are negatively linked to the provision of this ES. Ecosystem structure and processes and their interactions lead to functions. As for ES, the definition of ‘function’ is not unique. Going even further, Jax (2016) states that the existence of ‘function’ is problematic for ecologists and Wallace (2007) argues that the term function might even be unnecessary or to be avoided if processes, structure and composition are adequately defined. Taking an anthropocentric perspective, Potschin and Haines-Young (2016) state that there are advantages in thinking about functions, and refer to them as “taken to be the ‘subset’ characteristics or behaviours that an ecosystem has that determines or ‘underpins’ its usefulness for people”. Indeed, clearly identifying functions and the underlying structures and processes is helpful if we want to manage these properties in some way. Potschin and Haines-Young (2016) cite the case of woodland and their capacity to mediate runoff that can be controlled by their canopy characteristics which are not solely determined by woodland type. Similarly, the mediation of runoff function of forest ecosystems is supported by different processes (mainly evaporation, transpiration, infiltration, surface runoff) which influence this function in different ways. These processes are influenced by different “biophysical structures” (e.g. woodland type, vegetation density and structure, etc.). Trying to be clear about what capacities (properties, behaviours) make ecosystems useful to people, identifying these as 'functional' characteristics is therefore an important stage in understanding how ecosystems and people are linked.

If we continue downstream of the cascade, ES play a pivotal role being sort of the 'final outputs' of ecosystems leading to benefits that can be valued by people. Every single box and link defined in this cascade model brings its part of uncertainty, complexity and research needs. Moreover, as any model, this is a simplification of reality and the linearity and unidirectionality suggested in the cascade is a simplification of the complex reality. It is out of the scope of this PhD to elaborate or discuss each link or box but we will instead focus on the biophysical assessment of functions and/or ES, and the links between these two components. As argued below assessing this relationship is of particular importance for ES assessment and mapping exercises.

Moreover, this instability of the ES concept is reflected in the multiplicity of ES classifications and as Brauman et al. (2007) mention, the underlying conditions and processes in ES providing are so interlinked that "*any classification is inherently somewhat arbitrary*". Most of classifications group ES into three categories: provisioning, regulating and cultural services. In Europe, one reference classification is the Common International Classification of Ecosystem Services (CICES, see Table 1.1) developed from the work on environmental accounting undertaken by the European Environment Agency (Haines-Young and Potschin, 2010b, 2013).

Table 1-1. Common International Classification of Ecosystem Services (CICES), version 4.3, Jan 2013 (Haines-Young and Potschin, 2013)

Section	Division	Group
Provisioning	Nutrition	Biomass
		Water
	Materials	Biomass, Fibre
		Water
	Energy	Biomass-based energy sources
		Mechanical energy
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota
		Mediation by ecosystems
	Mediation of flows	Mass flows
		Liquid flows
		Gaseous / air flows
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection
		Pest and disease control
		Soil formation and composition
		Water conditions
		Atmospheric composition and climate regulation
Cultural	Physical and intellectual interactions with ecosystems and land-/seascapes [environmental settings]	Physical and experiential interactions
		Intellectual and representational interactions
	Spiritual, symbolic and other interactions with ecosystems and land-/seascapes [environmental settings]	Spiritual and/or emblematic
		Other cultural outputs

In CICES, the three widely accepted categories (named 'sections') are further divided into eight generic divisions, themselves subdivided in groups, which are further divided into classes themselves subdivided in class-types. The description of the service is progressively more specific when moving down the classification scale. Table 1.1 presents this classification limited to the division and group levels. The appropriate level can then be chosen depending on the purpose of the classification use, e.g. mapping, assessment or accounting information (Haines-Young and Potschin, 2013) .

1.2.3 Science and policy

History of the ES concept, definitions and multiple classifications developed in environmental accounting contexts contribute to its large use both in the scientific and the policy arenas. ES is often promoted as a means to bridge the gap between these communities in the context of addressing crucial challenges in land planning and ecosystems preservation. On the scientific side, the number of papers mentioning "ecosystem services" or "ecological services" rose exponentially during this Century first decade (Fisher et al., 2009). Schaich et al. (2010) counted over 2000 articles containing ES as a keyword in five of the most important journals including PNAS, Environmental Management, Biological Conservation, Ecological Economics and Ecology and Society. On the policy side, ES have been introduced in the programs of major international environmental NGOs like the World Wide Fund for Nature (WWF), the Wildlife Conservation Society (WCS), and the International Union for the Conservation of Nature (IUCN). The Convention on Biological Diversity (CBD) makes an explicit link between biodiversity and ES within its Strategic Plan for Biodiversity 2011–2020 (European Commission, 2011). In this regard, data and information on biodiversity supporting the implementation of the EU strategy and the Aichi targets in Europe are gathered on the Biodiversity Information System for Europe website (BISE, http://biodiversity.europa.eu/). The ES concept is also central in international initiatives such as the TEEB program (The Economics of Ecosystems and Biodiversity) or IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services). At the European level, the working group MAES (Mapping and Assessment of Ecosystem Services) has been established in response to the Aichi targets set up by the CBD [see Target 2, action 5 (European Commission, 2011)]. Assessment is defined in this context as the "*analysis and review of information for the purpose of helping* *someone in a position of responsibility to evaluate possible actions or think about a problem*" (Maes et al., 2013), revealing the diagnosis and help for management potential. At the national scale, several countries completed the assessment of their Ecosystems' states based on the ES concept such as the National Ecosystems Assessment of the UK (UK NEA, 2014) or Spain (Santos-Martín et al., 2014). Other activities are ongoing [see Schröter et al. (2016) for a review] such as in France with the "EFESE" program has been set up to assess Ecosystems and ES by biomes [French Assessment of Ecosystems and Ecosystem services (Puydarrieux, 2014)]. In Belgium, Ecosystems and ES trends have been assessed through the 'Flanders Regional Ecosystems Assessment' (Stevens et al., 2015) while in Wallonia a conceptual framework has been set up by the 'WalES platform' (Walloon Ecosystem services, http://www.wal-es.be/).

1.2.4 From science to practice: some points from the 'research needs hit list'

As mentioned, use of the ES concept is developing to bridge the gap between the scientific and policy spheres and several needs can be drawn from policy and scientific initiatives. De Groot et al. (2010) present several challenges to structurally integrate ecosystem services in landscape planning, management and design. Regarding the assessment of ES in particular, there is a need for assessing and discriminating between the 'supply' side of ES, i.e. the potential ES that ecosystems can deliver and the 'demand' side, i.e. the ES that are wanted by individuals and/or communities (Wolff et al., 2015). Furthermore, there is an important need for accurately quantifying every component of the ES 'cascade' through suitable indicators (Braat and de Groot, 2012; Burkhard and Maes, 2017; de Groot et al., 2010; Müller and Burkhard, 2012; Seppelt et al., 2011). This means accurately assessing the processes, functions, ES, benefits and their relationships. One of the main challenges in doing so is to deal with the high complexity of ecosystems' functioning and the complex dynamics characterizing the links between processes, functions, and services at different temporal and spatial scales (Bastian et al., 2012; Carvalho-Santos et al., 2014; de Groot et al., 2010; Swetnam et al., 2011; Turner and Daily, 2008; Villa et al., 2014). Assessing ES to support land planning decision-making remains thus a challenge due to multiple sources of uncertainty such as data scarcity, functional knowledge gaps, demand variability, etc. (Jacobs et al., 2013).

As a core part of spatially explicit assessment initiatives, ES mapping methods face similar issues. Mapping methods can be grouped into 6 broad categories: direct mapping, empirical models, simulation and process models, logical models, extrapolation (often based on land use and land cover (LULC) classes), data integration (Andrew et al., 2015; Englund et al., 2017). The first four types roughly constitute 'ecological production function methods' – implying the estimation of the level of ES provisioning at a particular location based on the biotic and abiotic characteristics of the site – while the latter two groups roughly constitute 'benefit transfer methods' where ES values are transferred from one context to another. These latter methods also called 'proxy methods' are often used in ES assessments (Albert et al., 2015; Egoh et al., 2012; Koschke et al., 2012; UK NEA, 2014). Arguably, the above-mentioned policies such as the EU biodiversity strategy 2020 targets (European Commission, 2011) provide an incentive for using 'proxy' techniques. Indeed, these methods, and in particular one of the extrapolation method known as the 'matrix approach' (Burkhard et al., 2010), allow for straightforward ES mapping.

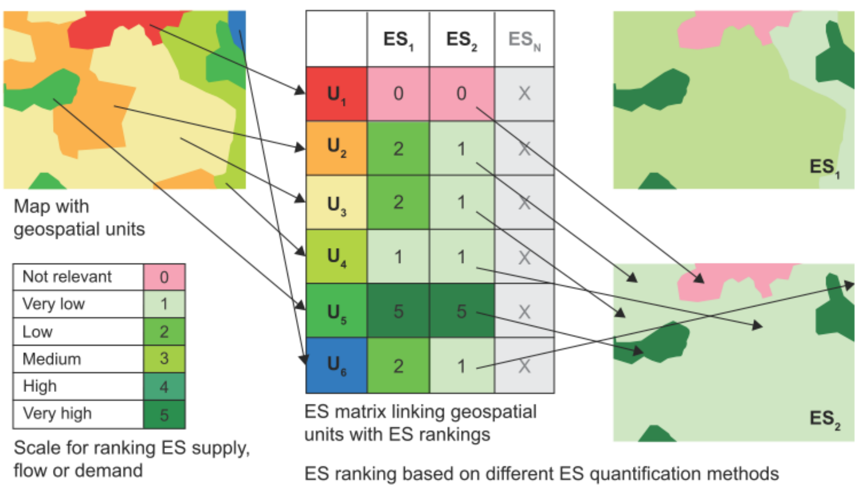


Figure 1-2. Overview of the ES matrix approach, based on geospatial map data, the actual matrix and resulting ES maps, source: Burkhard (2017)

In this so-called 'matrix approach' (Figure 1-2), ES are linked to appropriate geo-biophysical spatial units (Burkhard et al., 2010; 2017). Then, their supply and/or demand are ranked for each spatial unit in a pre-defined normalised scale (e.g. 0 not relevant to 5 very high), creating a 'matrix' that allocates for each spatial unit 'type' a provision (or demand) potential for each studied ES. This method has been largely applied and in particular, in its simplest form, i.e. when spatial units defined in the matrix are directly based on spatial LULC delineation. Indeed, the 'LULC-based matrix' approach can easily be implemented on large spatial extents and using common land use databases that are regularly updated (e.g. CORINE land cover, http://www.eea.europa.eu/publications/COR0-landcover). It is therefore often used in assessments and allows for comparison between countries. However, one may question the validity of these maps, as uncertainties are high and variable, in particular in terms of the expected direct and univocal links between LULC and the ES provided. Indeed, the same ES provision potential is allocated to every spatial unit from the same LULC class, no matter local characteristics (e.g. soil type, slope, management, etc.) leading to at best, unprecise, at worse false assessments.

1.2.5 Water related ecosystem services

Among major ES, water related ones are of prime importance to Humans. Indeed, water is the most essential component for the life of all beings, it is a major component of sustainable development and is crucial to healthy ecosystems, socio-economic development and to the survival of human beings (UN-Water, 2014; Haddadin, 2001; Falkenmark and Rockstrom, 2004). However, freshwater systems and consequently human well-being are directly threatened by human activities (Meybeck, 2003; Millennium Ecosystem Assessment, 2005a; Vörösmarty et al., 2010). Hydrological ecosystem services (HES), i.e. the benefits ecosystems supply by regulating the hydrological cycle (Willaarts et al., 2012), are therefore of prime importance. Brauman et al. (2007) broadly classified them into five categories including 'improvement of extractive water supply', 'improvement of instream water supply', 'water damage mitigation', 'provision of water-related cultural services', and 'water-associated supporting services'. These are delivered according to the following dimensions (further referred to as attributes) of: quantity, quality, location, and timing of flow (Brauman et al., 2007). Water quantity constitutes the amount of water available for drinking and non-drinking purposes or describes the volume of flood water. Water quality is a measure of the chemicals, pathogens, nutrients, salt and sediments in surface and ground water. Location means the location of delivery, while the timing attribute describes the moment when water is available. These HES attributes are directly impacted by ecosystems functioning, structure and management when water flows through the landscape. The water cycle encompasses water movements and its renewal. Its understanding and characterization provide important information to comprehend HES provision and the role of ecosystems in regulating water fluxes and composition. The water cycle and ecosystems interactions are illustrated in Figure 1-3. Arrows indicate fluxes of water. The water cycle, which is driven by solar energy, has no starting point. Processes affecting water fluxes are simultaneous. Part of surface water (mainly from oceans but also from rivers, lakes, etc.) evaporates and forms clouds along with water from terrestrial evapotranspiration. Then, this water falls as rain, fog, or snow (i.e. precipitations) onto Earth's land and oceans. On land, part of water infiltrates into the soil to groundwater or flows over the surface (surface runoff) or at the subsurface (subsurface runoff). In the local context, water and water fluxes are affected through diverse processes highly related to LULC: water use and interaction with vegetation, ground surface and soil modifications, local climate modifications, and water quality modifications.

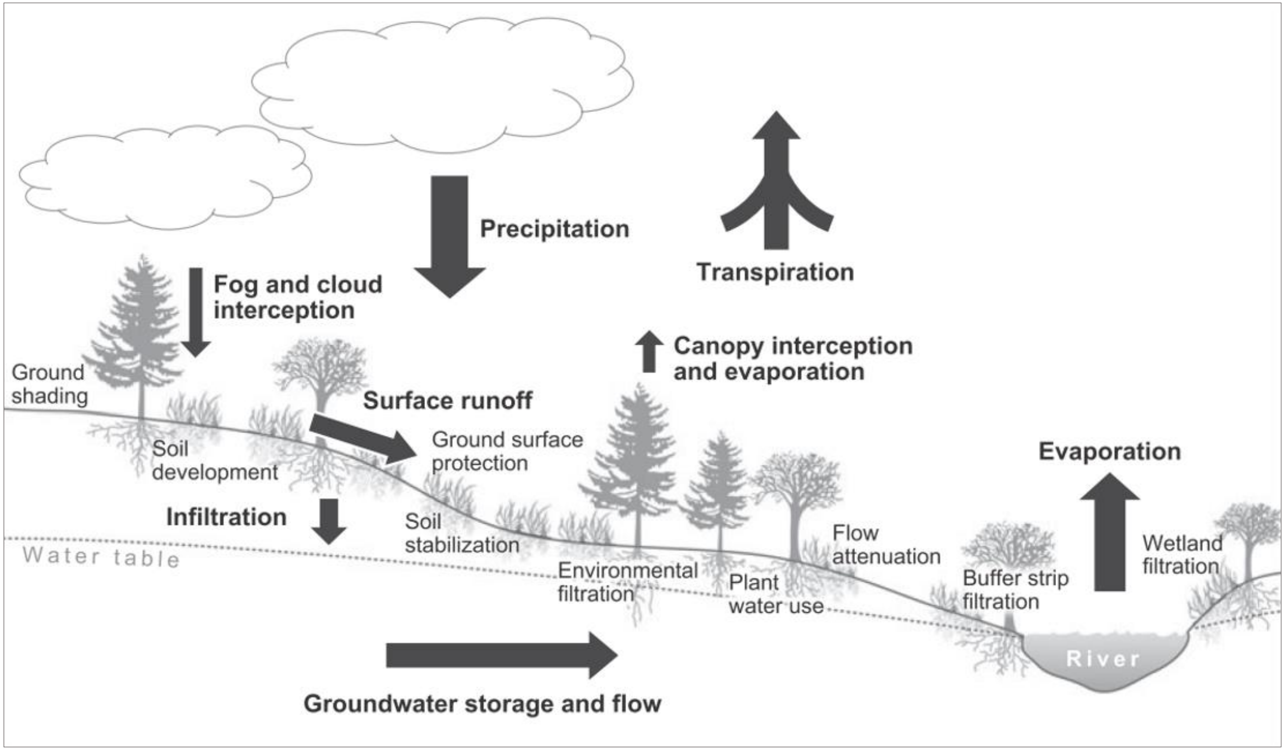


Figure 1-3. Water cycle flows and ecosystem interactions, source: (Brauman et al., 2007).

1.3 Forest cover impact on hydrological attributes and ecosystem services

Land cover and forests in particular affect HES through their impact on water and water cycle. Forests are seen as the main ecosystems interacting with water whether in terms of: quantity (i.e. total water yield), timing (i.e. seasonal distribution of flows) and quality (i.e. removal and breakdown of pollutants and trapping of sediments). Indeed, forests and forest soils alter each of the five physical, chemical and biological functions involving the reception, processing and transfer of water (Neary et al., 2009). This is mainly due to the following characteristics : (i) forests height, (ii) dense and irregular crown canopy with high leaf area index and lower albedo, (iii) architecture of their spread root system widely prospecting soil horizons, (iv) wide horizontal distribution and vertical coverage (Calder, 2002; Salemi et al., 2012; Zhang et al., 2001). Figure 1-4 presents an adaptation of the 'cascade' model in the context of forest HES based on Carvalho-Santos et al. (2014) framework. Mentioned services are those studied in the present PhD thesis based on Brauman et al. (2007) categories and the PhD's scope (supply side) is highlighted. The cascade model illustrates that forest ecosystems properties, constituted by biophysical structures (linked to plant structure but also to ecological variables) and processes, lead to functions. These functions contribute to HES characterized through attributes, themselves providing benefits to Human. These benefits can for example be linked to water supply to households (e.g. for direct consumption) or to primary or secondary sectors (e.g. to agriculture or industries). These benefits can also be related to the reduction in water bodies sediments content. Figure 1-4 illustrates HES and attributes studied in the present PhD (i.e. HES of instream water supply and water damage mitigation and attributes of quantity, quality and timing) and highlights our focus: the potential provision of HES by forests (supply side). Human management decisions result in pressures on ecosystems and actions to limit them. Main biophysical structures, processes and functions of forest ecosystems related to these HES are described based on literature review in sections 1.3.1 to 1.3.3.

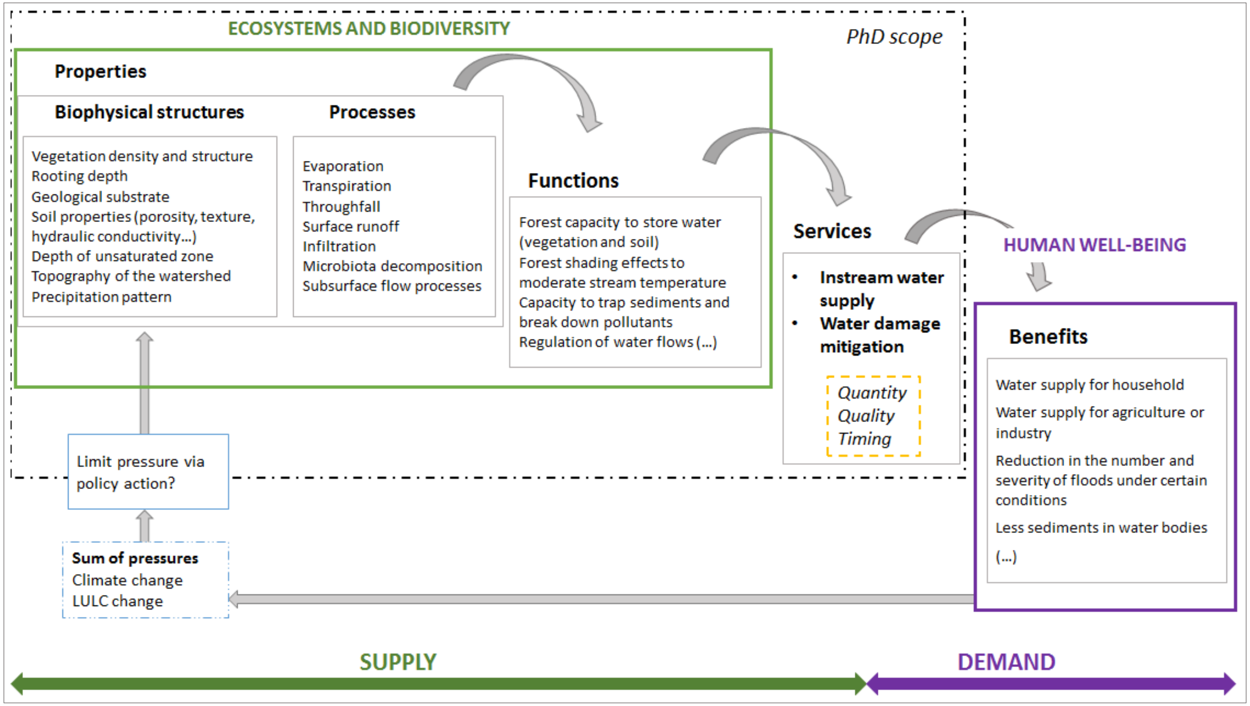


Figure 1-4. Cascade model for hydrological ecosystem services provided by forests adapted from Carvalho-Santos et al. (2014). Ecosystems properties (structures and processes) lead to functions leading to hydrological ecosystem services providing benefits to Human. Mentioned services studied in the present thesis are assessed in relation with their hydrological attributes: quantity, quality and timing. Human management decisions results in pressures and actions to limit them.

By explicitly listing operating processes and functions, Figure 1-4 highlights the complexity of water-related ES assessments. Indeed and as detailed in the following sections, different processes may be translated into several functions that impact the same HES in opposite ways.

1.3.1 Water related processes in forest

This section presents the main water related processes driving functions and HES delivered by forests in comparison to other ecosystems. The integrated effect of forest on water quantity, timing and quality and remaining gaps in the literature are detailed in sections 1.3.2 and 0.

One of the main processes acting on water fluxes distribution in forest is **evapotranspiration (ET),** which describes the total loss of water as vapour from the biosphere. This includes water vapour lost through interception by the canopy, through evaporation from the soil surface, and transpiration from trees and understory, the latter flux being regulated by the species according to their water stress tolerance.

Interception is the first process resulting from the interaction of forest and precipitation. It represents the fraction of water that is evaporated back to the atmosphere from the canopy (or absorbed by leaves) and that never reaches the forest floor. Interception values are highly variable according to the tree species, sylviculture, climate and season. In temperate climate, interception represents from 15 to 45% of incident rainfall (Nisbet, 2005; Office National des Forêts, 1999). Needle-leaved forests have higher interception rates [from 25 to 45% (Nisbet, 2005; Office National des Forêts, 1999)] than broad-leaved forests (from 15 to 30% (Office National des Forêts, 1999) or 10 to 20% according to Nisbet (2005)). This is mainly due to (i) higher leaf area index (LAI) values for needle-leaved species and (ii) their evergreen character (except for *Larix sp.* and other non-evergreen needle-leaved species). Interception rates are negligible for grassland or arable crops while they present similar values to broad-leaved forest when considering heather or bracken (Nisbet, 2005).

Transpiration (T) is the process by which water taken in by plant roots from the soil is evaporated (i.e. loss) through the pores or stomata on the surface of leaves. Tree transpiration is often the main component of the evapotranspiration flux. In the absence of water stress, it is directly driven by the energy received at the canopy that can be quantified by the potential evapotranspiration (PET), and the T/PET ratio can be as high as 0.8. Broad-leaved forests transpire more than needle-leaved species during the vegetative period (in Belgium, from around April to September) but on an annual basis, transpiration rates are similar (from 30 to 35% of received rainfall for needle-leaved species and from 30 to 39% for broad-leaved species according to Office National des Forêts (1999) and Nisbet (2005) respectively).

Evaporation from the soil and the transpiration from understory vegetation complete the ET term. These fluxes are highly variable, depending on climate, canopy cover and species composition. According to some authors (Daikoku et al., 2008; Osberg, 1986; Wilson et al., 2000) soil evaporation can represent 15 to 21 % of a stand's ET (with no herb layer). Average summer transpiration from understory vegetation can represent 34% of a stand's ET (Gobin, 2014).

Different techniques can be used to quantify ET and they differ a lot regarding the spatial and temporal study scales. ET at the catchment scale can be derived from the measurements of individual fluxes (i.e. transpiration, interception and evaporation) at a really local scale (leaf, tree) that need to be then upscaled to the stand and catchment scales with associated high uncertainties (Ford et al., 2007). Eddy covariance method provide ET assessment through water vapour fluxes measurements above canopy at a larger spatial scale than the tree (i.e. a portion of stand) but variable in time (see e.g. Aubinet (2001)et al. and Soubie et al. (2016) studies in Wallonia, Belgium). Another widely used approach is based on the water balance study at larger spatial and temporal scales. The water balance builds on the principle of mass conservation and states that the 'inflow' minus 'outflow' equals the variation in water storage (Office National des Forêts, 1999) in the system. One of the main spatial scale at which the water balance is studied is the catchment scale which can be defined an area hydrologically closed, where no flow is coming from the exterior and where the overflow issued from rainfall either evaporate or flow to one unique section. If we consider the catchment scale, the water balance equation can be written as follows:

$dS/dT = P - Q - ET$ [Equation 1.1]

with S : catchment water storage (m^3), P : Precipitation (m^3/s), Q : discharge the catchment outlet (m^3/s), ET : actual Evapotranspiration (m^3/s). Studying the water balance at the catchment scale allows thus to derive information on the forest cover action (ET term) if other terms are known (P and Q) and differences in catchment water storage neglected (over adequate temporal frames). Discharge at the catchment outlet appears thus as a key variable, easily and frequently monitored, reflecting and integrating the processes partitioning water fluxes.

Studies show that annual ET varies greatly between biomes, according to climate, soil characteristics (mainly depth, texture, organic content and slope) and type of forests: ET/P may vary between 0.25 up to 0.85 (Larcher, 2003; Vose et al., 2011). As abovementioned, the partitioning of the ET fluxes also varies greatly with forest composition and phenology, density, age, structure, and therefore strongly impacts the amount of rainfall reaching the soil, the amount of soil water available and forest productivity. However, **forest is acknowledged to have a higher evapotranspiration (ET) rate than lower vegetation** as grass or arable land (Amatya et al., 2016; Granier, 2007; Office National des Forêts, 1999; Zhang et al., 2001). Also, regarding difference between forest types, we can assume that **needle-leaved tree species have higher annual evapotranspiration rates than broad-leaved tree species** (Nisbet, 2005) mainly given their higher interception rates.

Throughfall, defined as water that reaches the soil either directly through canopy gaps or indirectly after running off the canopy, represents from 60 to 90% of rainfall. **Stemflow,** defined as water running down the trunk and into the soil represents generally small values (1 to 4% of rainfall) (Williams, 2016) but can reach in some cases (i.e. for beech) values as high as 12-18% of rainfall (Barbier et al., 2009). Throughfall and stemflow represent the fraction of incident rainfall which is not intercepted by the canopy and re-evaporated. These are obviously also variable according to tree species, size, density and canopy roughness (Williams, 2016), see eg. Barbier et al. (2009) for a review of the influence of several tree traits on rainfall partitioning in temperate and boreal forests.

Water that reaches the soil and can either infiltrate into the soil or runoff on the surface or in the subsurface. The partitioning of these fluxes is complex and will depend on the soil characteristics and pre-existent water content.

Infiltration represents the water movement under gravity and pressures entering the upper soil layers towards the subsoil. A part of this water contributes to rise soil water content**,** another part **percolates** to ground water while another part reaches the stream quicker. Authors argue that **infiltration rates are higher in forest mainly due to forest soils higher porosity, litter and presence of canopy** which slows down rain drops (Bruijnzeel, 2004; Calder, 2002).

Le Maitre et al. (2014) proposes a comparison of schematic representations of components and processes linked to flow regulation service, under "well-managed" and "modified" ecosystems (Figure 1-5). This figure illustrates how the components and processes that control the water partitioning in the ecosystem (i.e. interception, evaporation, transpiration, infiltration, percolation, surface and subsurface runoff) are affected under inappropriate land use. Considering effects of forest on above-mentioned processes partitioning rainfall; clear-cutting, thinning, sanitation problems will impact this partitioning and be reflected in streamflow (i.e. discharge measures).

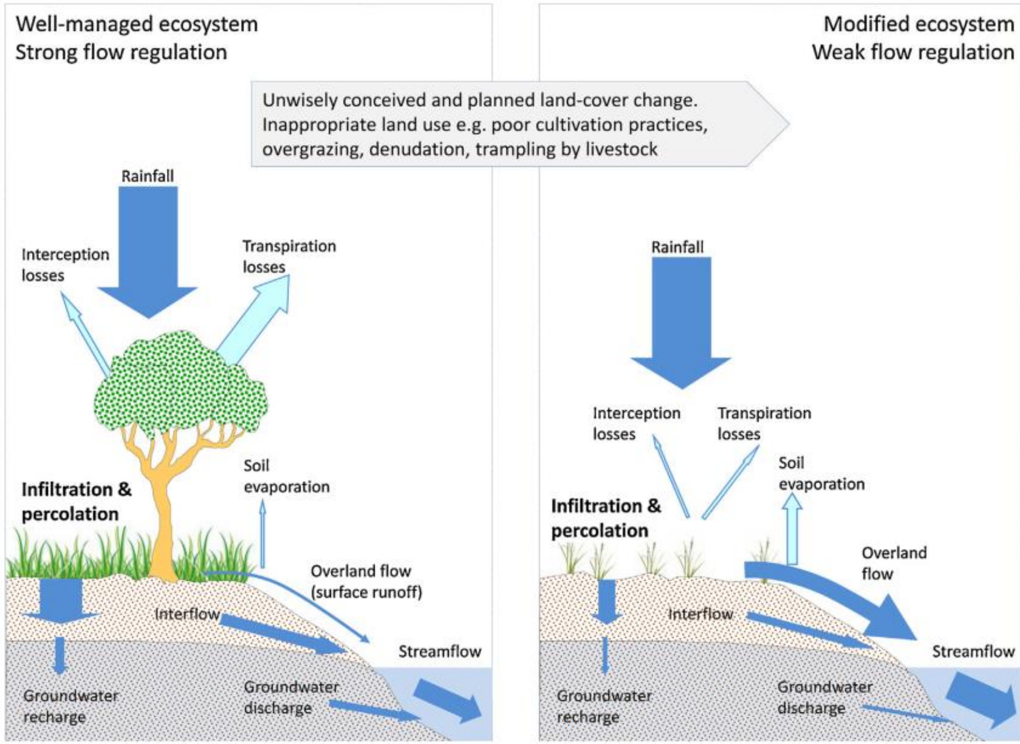


Figure 1-5. The movement of water from precipitation through the vegetation and soil system into streams, and how it is affected by changes in vegetation, from Le Maître, et al., (2014).

Biological, chemical and physical processes altering the composition of water occur in forest ecosystems. Compartments of canopy and floor in these ecosystems play an important role in nutrient cycling, stream water chemistry, and stream water quality (Arocena, 2000). From the physical point of view, forests, more than other vegetation types, minimize soil erosion on site and reduce the amount of sediments in water (wetlands, ponds, lakes, streams, rivers). Studies also showed that forest trap or filter water pollutants in the forest litter (Calder et al., 2007). Indeed, forest litter provides a physical barrier to splash-induced erosion. Forest surface cover, debris and tree roots trap sediments and deep tree roots stabilize slopes. Pollutant removal processes within forest and riparian buffer can occur at the surface or subsurface level. Pollutants and particles removed from the surface are sediments and nutrients such as phosphorus, trapped by grasses brush and shrubs. Subsurface pollutants removal such as groundwater NO3--N can occur through plant uptake or denitrification process by microbiota (Madigan et al., 2014). Nitrate removal by riparian buffer is one of the most studied process regarding water quality (see e.g. Sabater et al. (2003) study concerning this process across a climatic gradient in Europe) and testifies an "active" effect of forest on improving water quality.

This section aimed at (i) presenting main processes related to forest cover effect on water and (ii) providing an order of magnitude of these in comparison with other vegetation types. The following sections present knowledge about the integrated effect of these processes in forest on HES and their attributes of quantity, timing and quality.

1.3.2 Water quantity and timing

To study the effect of forest cover on the water quantity and quality attributes at the landscape scale, many studies adopt a 'catchment' scale approach, as it appears to be a relevant spatial unit of study because of its integrative character (Granier, 2007). Indeed, studies that focus on measuring precisely the water cycle fluxes at the stand scale exist but it is highly challenging to upscale them (Asbjornsen et al., 2011; Oishi et al., 2008; Schume et al., 2003; Schwärzel et al., 2009; Unsworth et al., 2004; Vincke et al., 2005; Wilson et al., 2001).

As already mentioned when describing hydrological processes, various factors impact these, resulting in high variability when quantifying them.

Despite the many studies conducted to measure the impact of forest cover on water cycle components (Brown et al., 2005; Farley et al., 2005; Robinson et al., 2003), relationships between water flows (quantity and timing) and forests have been controversial since Pliny the Elder (Andréassian, 2004b). Many of these studies are paired-catchment studies [see Bosch and Hewlett (1982) for a review] where catchment size is often small. Indeed, studied catchment areas cover for the vast majority less than a few km^2 limited by the fact that these experiments require controlling most of the factors impacting water flows while having pure and distinct land covers between catchments. These studies present the major disadvantage to lack experimental replications across a full range of natural conditions (DeFries and Eshleman, 2004). Regarding annual water yield, results indicate an increase of it when forest cover is replaced by lower vegetation cover. More precisely, Sahin and Hall (1996) report from an analysis of 145 experiments that for a 10% reduction in cover of coniferous or eucalyptus (which is a high water demanding broad-leaved tree species), water yield increased by 20-25 mm and 6 mm respectively. This can be explained by higher evapotranspiration rates found in forests vs other LULC and higher annual evapotranspiration rates in needle-leaved forests vs broad-leaved forests. This reduction in water yield may negatively affect the instream water supply but could favour the water damage protection service (Brauman et al., 2007). On the other hand, at the global scale, authors argue that forest cover raises the precipitation events likelihood and increases water yield by contributing to the availability of atmospheric moisture vapour and the transport across continents (see Ellison et al. (2012) for a "forest-water yield" debate review). In addition to contribution to atmospheric moisture, forests could also contribute to increase available water by favouring infiltration (van Dijk and Keenan, 2007).

Other debates and perceptions confrontation are taking place in the popular and scientific spheres regarding the capacity of forest to regulate the timing of flows (i.e. their seasonal distribution) and in particular to reduce peak flows and promote water availability in low flows (Calder, 2002; Ceci and FAO, 2013). Global consensus on positive or negative effect on regulation of peak and low flow are hardly found in literature given the variability between sites linked to climate, soil characteristics, type of forests, etc. However, there is a partial consensus growing in the literature stating that forests may help to mitigate floods from small storms; but this role is variable according to geology, soil and climatic conditions (Cosandey et al., 2004). This positive effect of forest on flood protection tend to decrease with the intensity of storm and the pre-existing high soil moisture condition (Calder and Aylward, 2006; Lana-Renault et al., 2011).

Regarding low flows and seasonal distribution of water the same trade-off as for water yield exist, between a higher evapotranspiration versus an assumed favoured infiltration capability and an increase in atmospheric moisture by forest ecosystems. During summer, we can expect higher transpiration losses because of the deeper root systems of trees, reducing soil water reserves which sustain base flows (Calder, 1992). Differences in forest type effect can also occur with regard to differences in evapotranspiration rates according to phenology. Broadleaved forests have a higher evapotranspiration than needle-leaved forest in summer (essentially due to higher interception). However, univocal conclusion regarding the effect of forest cover on dry season flows in comparison to lower vegetation can not be drawn from the literature (Calder, 2002). Effects on dry season are likely to be site specific.

Finally, global changes push scientists to claim for renewing studies linking hydrological processes and land cover. DeFries and Eshleman (2004) claim for research about interactions between land-use change and hydrologic processes, stating that it is and will be a major issue in the decades ahead. More generally, Vose et al. (2011) state that global changes which affect water quality and quantity (i.e. climate change, land use change and invasive species) question the assumption that studies from the last decades can be used to face future conditions. Indeed, these authors state that the intensification of human activities across the globe have created conditions that are outside the range of many of our historical observations and understanding derived from those observations. In particular, climate warming will likely result in increases in evaporation and more intense precipitation events leading to the hypothesis that one of the major consequences will be an intensification (or acceleration) of the water cycle (Del Genfo et al., 1991; Huntington, 2010; Loaiciga et al., 1996) along with a general exacerbation of extreme hydrologic anomalies such as floods and droughts (Easterling et al., 2000; Gleick, 1989).

1.3.3 **Water quality**

Water quality management is at the core of policies such as the US Clean Water Act (1972) and the European Water Framework Directive (Directive, 2000/60/CE) which share the common objective to maintain or restore the chemical, physical and biological integrity of surface waters. Water quality can be described by hundreds of variables which can broadly be classified into physical, chemical and biological categories (Boyd, 2015; Chapman, 1992). These groups of variables provide complementary information and are inter-related.

Managing water quality is challenging and implies to deal with both point and non-point source pollutions. Land use and land cover are key landscape elements affecting water quality through their impact on non-point source pollution resulting from complex runoff and landscape interactions. Giri and Qiu (2016) stress the importance of assessing the relationship between LULC and water quality. To their point of view, improving the understanding of these relationships can help managing water quality in unmonitored watersheds but also providing recommendations to watershed managers and policymakers in the land planning decision process. In order to capture forest cover impact on water quality and avoid driving simplistic conclusions, one must consider other LULC notably those associated with pressures on water quality. Negative impact of agricultural intensification is reported in the literature (Stoate et al., 2001) mainly explained by the following processes: increased sedimentation, modified hydrological regimes, loss of high quality habitat, contamination from pesticides, increases in surface water nutrients (mainly N and P) (Allan, 2004; de Oliveira et al., 2016; Herringshaw et al., 2011; Mahler and Barber, 2017). Urban land use and urban intensification are also reported to negatively affect water quality (Kosuth et al., 2010; Riva-Murray et al., 2002; Yu et al., 2013). Forest, on the contrary, is usually associated with water containing less sediments and fewer nutrients (Neary et al., 2009; TEEB, 2010). Some studies showed positive impact of forest cover on instream water quality (Kosuth et al., 2010; Tong and Chen, 2002). This positive effect is likely due to both an “active” effect related to the active trapping or filtering of water pollutants and a “passive” effect being linked to the absence of more polluting practices associated with other LULC (e.g. agricultural cover). Indeed forestry activities generally use no fertilizers or pesticides.

Spatial location of LULC and in particular forest cover raises questions with regard to its impact on water quality. The overall catchment cover and that of the riparian zone (defined by Naiman et al. (2005) as "transitional semi-terrestrial areas regularly influenced by fresh water, normally extending from the edges of water bodies to the edges of upland communities") are often studied. However, the question addressing the scale at which land use within stream catchments most influences stream water quality and ecosystem health remains only partially answered (Allan, 2004; Johnson et al., 1997; Sheldon et al., 2012; Sponseller et al., 2001). Several studies suggest that prevailing (Kail et al., 2012; Riva-Murray et al., 2002) and past (Harding et al., 1998) LULC characteristics of stream catchments affect surface water quality. Other studies emphasize the impact of riparian LULC on water quality or stream habitat (Dosskey et al., 2010; Jackson et al., 2015). Finally, some studies compare scales of influence (i.e. catchment scale versus riparian scales), obtaining nuanced results on the land use effect on stream water quality according notably to the type of biological indicators and the ecological context of the sampling sites (Kosuth et al., 2010; Marzin et al., 2012, 2012; Sponseller et al., 2001). Specifically, forested riparian buffer zones are believed to have a positive impact on water quality through notably the reduction of the nutrient concentrations in water (Dosskey et al., 2010; Fernandes et al., 2014; Naiman et al., 2005; Scarsbrook and Halliday, 1999). However, this is nuanced by studies explicitly assessing the effect of riparian forest compared to catchment forest (Stephenson and Morin, 2009). These studies show that assessing both scales of influence bring deeper insights when studying LULC impact on water quality (Vondracek et al., 2005).

1.4 Preliminary conclusions

Given the above, we may highlight the following conclusions and associated caveats in the literature with regards to assessing water related ES in the broader context of ES assessments. ES concept and frameworks such as the cascade model, based on scientific literature, acknowledge the Human dependency towards Nature and more specifically the links between ecosystems structure and processes, function, ES and human well-being. In doing so, one of this concept's purposes is to act as a tool for better resources assessment and management in a context of degradation of ecosystems and their ES. However, there is a **crucial need for accurately quantifying every component of the ES 'cascade'** through suitable indicators accounting for the complexity of these relationships. While current policy-driven initiatives of ES assessments and mapping are often based on methods relying on simple land cover proxies (such as the above-presented 'LULC-based matrix' approach), **research is needed to validate or not these proxies and when appropriate, propose indicators that can easily be mapped, but better reflect the underlying processes underpinning ES supply**.

Among ES, those related to water are of prime importance and known to be influenced by ecosystems and LULC. Regarding forest cover, its associated assumed effects on processes related to water quantity are a high evapotranspiration, high infiltration compared to surface runoff or rapid drainage (at least on low slopes), increase of soil moisture content, recharge of groundwater and the gradual release of water. However, the **combined effect of these processes on hydrological ES is less evident to derive given the ecosystem complexity and heterogeneity at the landscape scale.** Regarding LULC effect on water quality, there is an opposition between forest cover associated with higher water quality and agricultural and urban land associated with lower water quality. Forest cover processes result in water with less sediments and nutrients. However, **questions related to the integrated effect of mixed LULC at the landscape scale and regarding the forest position in the landscape** (i.e. within riparian zone or whole catchment) **where its effect on HES is the strongest remain unanswered**. This appears to be relevant in relation with policies such as the European Water Framework Directive [EU-WFD, (European Commission, 2000)]. Finally, global changes push for renewing of these studies linked to effect of ecosystems on HES.

1.5 PhD thesis objectives and scope

1.5.1 Thematic and methodological objectives

In an attempt to tackle these research gaps, the **focal objective** of this PhD thesis is to assess the **impact of forest cover on hydrological ecosystem services.** The HES studied in this PhD thesis are the instream water supply and the water damage mitigation service of flood protection. This study focus therefore on river flows. It also clearly focus on the supply side of the cascade model (see Figure 1-4) representing the potential provision of these services by forest cover *versus* other LULC.

The main objective is declined in three sub-objectives and associated with methodological objectives (see Figure 1-6) which are described below.

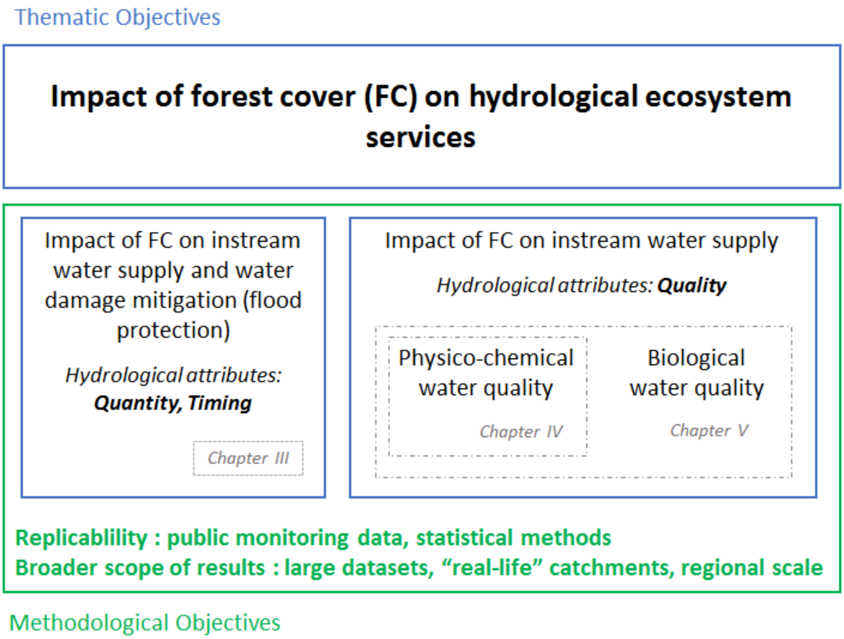


Figure 1-6. Objectives of the study (blue: thematic objectives; green: methodological objectives) and corresponding chapter numbers.

In order to fulfil the main objective, **transversal methodological objectives** are pursued:

(i) to ensure replicability of the methods,

(ii) to broaden the scope of the results, moving towards land planning oriented results.

The main thematic objective is declined and completed through three specific objectives, related to the study of the impact of forest cover on:

(i) instream water supply and the water damage mitigation service of flood protection. In this study, the hydrological attributes of quantity and timing are studied in the Ardenne ecoregion (Chapter 3)

(ii) instream water supply in terms of physico-chemical water quality at the regional scale (Chapter 4)
(iii) instream water supply in terms of biological water quality at the regional and subregional scale (Chapter 5)

More specifically, in Chapter 3, the independent effect of forest cover types (i.e. needle-leaved and broadleaved forests) on instream water supply and flood protection is assessed.

In Chapter 4, the independent effect of forest cover types (i.e. needle-leaved and broadleaved forests) on physico-chemical water quality in comparison to other LULC is quantified at the regional scale. The link between sub-catchments' LULC and the legal status of in stream water quality is analysed. Furthermore, the annual and seasonal effects on the forest cover impact on physico-chemical water quality are assessed.

In Chapter 5, the forest cover effect on biological water quality in comparison to anthropogenic pressures and physico-chemical water quality is quantified. Furthermore, we compare the effect of riparian forest to the proportion of forest in the upstream catchment. The effects of population density and local morphology on the forest cover effect on water biological quality are also assessed. The study scale is twofold: regional scale (Wallonia) and subregional ecoregion scale.

Figure 1-7 presents the main objectives, specific objectives, and the scales of study.

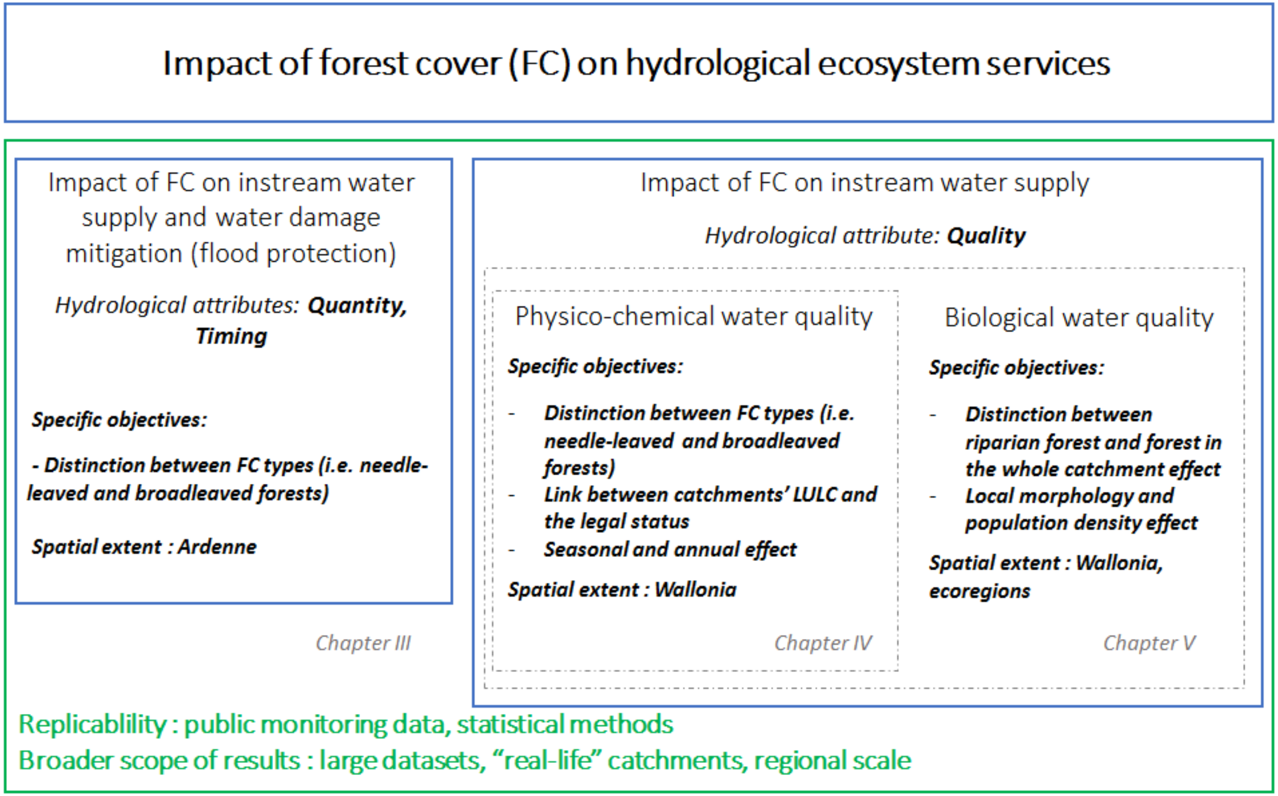


Figure 1-7. Specific objectives and spatial extents of the studies (blue: thematic objectives; green: methodological objectives).

1.5.2 PhD scope and work assumptions

The interdisciplinary character of this PhD is reflected in its objectives – be it thematic or methodological – and the methods needed to fulfil them. Our contribution first fits into the interdisciplinary nature of the **ecosystem services approach**. More specifically, we aim at bringing insights to the biophysical assessment of HES from the 'supply' point of view (Figure 1-4).). In doing so, we also aim to enrich the debate of using land cover proxies (see Figure 1-2 and section 1.2.4) versus more advanced methodologies, such as detailed and precise simulation and process based models or direct mapping. This may contribute to deriving indicators used to map water related ES at the landscape scale, yet meaningful in land planning processes.

Second, our work aims at bringing information and answering questions with regard to **ecohydrology** science and its present and future challenges (Vose et al., 2011). However, given our objectives and, in particular, our aim to contribute to better land planning processes, the spatial scale of study is completely different from that of traditional ecohydrologists studying regulation of fluxes. Indeed, we aim at bringing insights on forest cover effect on water at the landscape scale where ecohydrological processes are highly relevant for society through their impacts on water provisioning and quality (Asbjornsen et al., 2011).

Regarding this context, several work assumptions and choices support the present study.

First, the **'catchment' scale** appears to be a relevant spatial unit of study because of its integrative character (Granier, 2007) reflecting effects of accumulated fluxes. These accumulated fluxes resulting from complex ecosystems interaction with water are manifested by streamflow, evapotranspiration and recharge. Discharge at the catchment outlet appears therefore as a key variable reflecting LULC and in particular forest cover effects on water fluxes partitioning while being an easily and frequently monitored variable. Indicators chosen to describe the quantity and timing dimensions of HES will therefore be based on discharge series (Chapter 3). Similarly, indicators describing water quality will also be measurements at the catchment outlet (Chapters 4 and 5).

In order to broaden the scope of the results obtained and to derive land planning oriented results, our study extent varies from ecoregional to regional scale. We study “real-life” catchments (vs. small controlled pure LULC catchments), ranging from a few to hundreds km^2 with mixed land covers and a specific focus on forest cover. This is directly induced by the spatial heterogeneity of LULC in the study area at the landscape scale, as in many other countries worldwide. In this context, “**forest cover**” is studied through a **proportion of forest cover in upstream catchments**. This implies that we study various forests in terms of management, stand age, tree density, species combination, local conditions. In Chapters 3 and 4, forest cover type effect is studied based essentially on foliar phenology by discriminating between needle-leaved forest and broadleaved forest covers. Also, spatial extents of studies are relatively large, from one ecoregion to the regional scale, multiplying sites condition cases (from geological, soil, climatic, tree species, associated LULC, management points of view). This study has indeed an inductive character, as we aim at deriving trends and explanations from large and variable datasets. In order to cover several climatic configurations (regarding rainfall and temperature), while overall representative of the study area, we worked with as much as data as possible. Regarding water quantity, timing and physico-chemical water quality, given the available datasets (i.e. discharges, rainfall and physico-chemical datasets), we chose to study 10 years of data (2005-2014) (Chapters 3 and 4). Regarding biological water quality (Chapter 5), we worked with the last EU-WFD cycle data (i.e. 2009-2014, as data from 2015 were not validated yet).

To ensure replicability, developed methods take advantage of public data monitored in many countries [whether regarding water quantity and timing (discharges) or water quality (monitoring measures in the EU-WFD framework)]. Given (i) the spatial scale and extents of study, (ii) the large number, the high diversity and heterogeneity of studied catchments, and (iii) our objective to study the integrative effect of forest cover on water related HES rather than its effect on individual processes, we develop robust and relatively simple **statistical methods** (see, in particular, section 2.3).

Chapter 2 Material and Methods

The present section aims at briefly presenting the study area and the datasets used in every chapter. Some methodological choices are also briefly presented as Chapters 3, 4 and 5 present each piece of study in detail, including a "Material and Method" section.

2.1 Study Area

The study area is the southern region of Belgium (Wallonia) covering 16 902 km^2 (ca. 55% of Belgium's area, see Figure 2-1 A). Wallonia presents relatively contrasted landscapes and can be divided into five natural ecoregions (Figure 2-1 A and Table 2-1). Noirfalise (1988) delineated these ecoregions according to pedological, botanical and agro-ecological criteria. Table 2-1 presents their main characteristics regarding LULC, topography, and rainfall distributions. More specifically, from north to south:

The **Loam region** is a low plateau covered with a thick silt layer presenting the Walloon region's mildest making it highly suitable for arable crops and grasslands (69%). This ecoregion is made of open valleys with gentle slopes with high anthropogenic pressures (intensive agriculture, high population density,…) and a low forest cover proportion (10%). Rivers and riparian areas are particularly degraded and modified due these anthropogenic pressures but also to the natural landscape configuration (i.e. gentle slopes and loamy substrate) limiting the natural restoration processes capacity (Michez et al., 2017).

The **Condroz** is a plateau presenting a steeper relief than the Loam region, located in the south of the Sambre and Meuse rivers. It presents a particular topography formed by the spatial alternation of psammitic rocks on the crest lines mostly covered in forest and limestone in the valleys where grassland and cropland covers are found. The average population density is high and strongly influenced by the presence of three nearby cities (Charleroi, Namur and Liège). Agricultural activity is less widespread than in the Loam region but locally important and forest cover is moderate (24%).

The **Famenne** is a large depression with an impervious and shallow soil layer composed of clay and schist. To the south of this ecoregion, a narrow limestone strip outcrops locally: the ‘Calestienne’. Grassland is the main land cover (36%) and the population density is quite low. Forest cover represent a third of the total area.

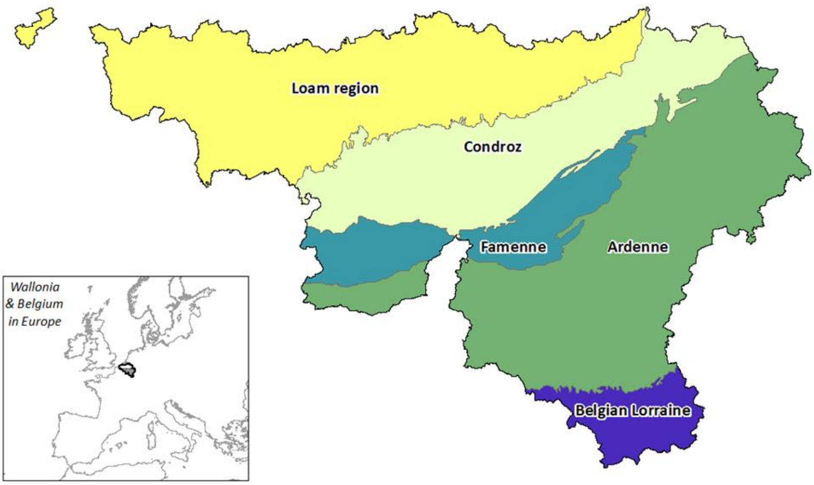
The **Ardenne** is a high plateau dissected by several rivers constituting the western protruding end of the “Rhine great schistose massif”. This region presents the highest elevation zones (with a highest point at 694m, at the “signal de Botrange”). The climate presents a continental character and is on average the rainiest and coldest in the country. This ecoregion has a low population density (44 inhab/ km^2) and a high forest cover proportion (56%).

The **Belgian Lorraine** is highly contrasted region in the South of Wallonia. Geologically, it is constituted by a succession of three “cuestas” oriented west-east. In terms of land cover it presents both agricultural zones (mainly grassland) and forested zones.

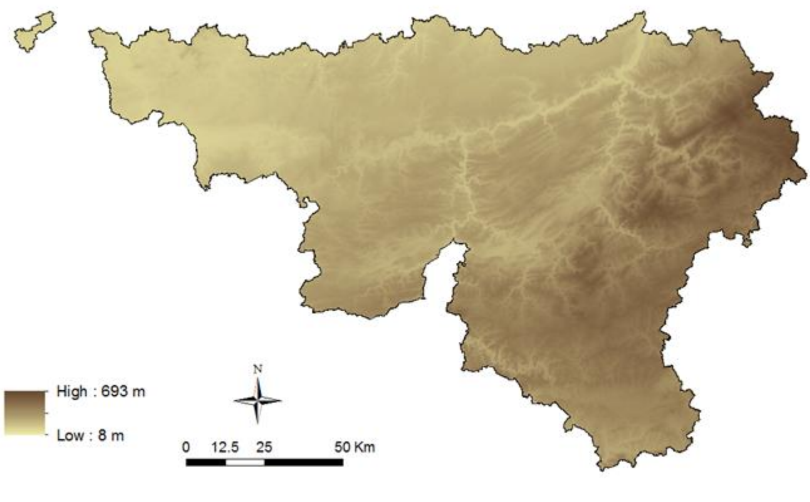
Table 2-1. Main ecological characteristics of Wallonia and its ecoregions, with Cr.: cropland cover, Gr.: grassland cover, Urb. : urban LULC, For. Forest cover, Wat.: water, source LULC: Top10VGIS).

	Area (km²)	**Rainfall** (mm / year)	**Mean altitude** (m)	**Mean slope** (%)	**Cr.** (%)	**Gr.** (%)	**Urb.** (%)	**For.** (%)	**Wat.** (%)	**Mean pop density**
Loam region	5192	825	103	4.8	51.2	17.5	19.2	10.3	<1	320
Condroz	3570	956	214	9.8	25.1	29.5	18.7	24.5	1.2	344
Famenne	1574	898	227	9.3	12.1	35.8	9.0	41.4	1.0	74
Ardenne	5710	1140	425	11	5.0	29.2	7.1	56.3	<1	44
Belgian Lorraine	851	934	322	9.1	12.6	33.7	10.3	41.6	<1	107
Wallonia	**16898**	**971**	**258**	**8.5**	**24.5**	**26.5**	**13.6**	**33.3**	**<1**	**208**

A. ECOREGIONS



B. ELEVATION



C. MONITORING STATIONS & HYDROGRAPHY

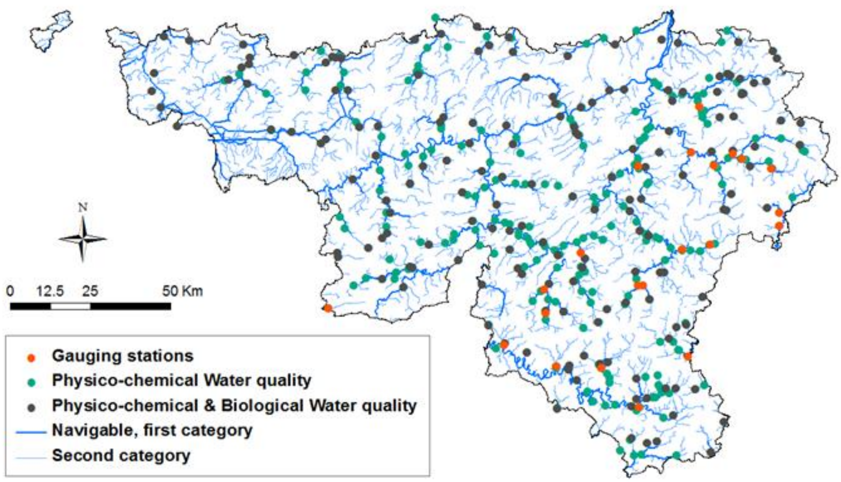


Figure 2-1. (A) Ecoregions in Wallonia;(B) Elevation (source: regional LiDAR digital terrain model, http://geoportail.wallonie.be)), (C) Hydrography, gauging stations (Aqualim network, Chapter 3), Physico-chemical water quality monitoring stations (EU-WFD monitoring network, Chapters 4 and 5), Physico-chemical water quality monitoring stations (EU-WFD monitoring network, Chapter 4)

Figure 2-1 B illustrates the elevation in Wallonia while Figure 2-1 C presents the monitoring stations of this study. In particular, orange dots represent the gauging stations studied in Chapter 3, grey and green dots, the physico-chemical water quality monitoring stations studied in Chapter 4 and green dots the physico-chemical & biological water quality monitoring stations studied in Chapter 5.

<u>*Forest cover*</u>

Most of the forest cover in Wallonia is represented by either needle-leaved (44%) or broad-leaved forests (53%), the rest being classified as mixed forest (3%) (source: Top10VGIS). Needle-leaved forests – mainly located in the Ardenne – are intensively managed with the use of exotic species (mainly spruce (*Picea abies*) but also Douglas fir (*Pseudotsuga menziesii*), larches (*Larix* sp.), and pines (*Pinus sylvestris* and *P. nigra*)). These are conducted in even-aged stands with systematically clear-cuttings, and high drainage infrastructure when located on wet soils. Broad-leaved forests – which, in contrast with needle-leaved forests, spread across Wallonia – are largely dominated by oaks (*Quercus robur* and *Q. petraea*) and beech (*Fagus sylvatica*). Other species such as birch (*Betula pendula*), ash (*Fraxinus excelsior*), maple (*Acer pseudoplatanus*), and hornbeam (*Carpinus betulus*) are also present (Alderweireld et al., 2015).

2.2 Variables of study and databases

Public monitoring networks data from the Walloon Public Service (WPS) were used to describe water related attributes and HES. The following sections detail the hydrological variables and dataset used in Chapter 3 and the water quality variables and datasets used in Chapter 4 and 5. Finally, the LULC dataset, common to every study, is presented.

2.2.1 Water quantity and timing: hydrological variables

In order to study the impact of forest cover on water quantity and timing (Chapter 3), we based our study on the water balance approach, which is itself based on the principle of continuity. As discharge at the catchment outlet is reflecting and integrating the processes partitioning water fluxes, we derived hydrological indicators (see Chapter 3) from daily discharge measurements. Indicators derived from instream flow (e.g. annual water flow) are indicators of the "capacity to store water" ecosystem function (see Figure 1-4) (Carvalho-Santos et al., 2014; Maes, 2011) that can be directly linked to the instream water supply service (Garmendia et al., 2012). We approached the flood protection service through other variables derived from the discharge data series: the specific discharge exceeded 5% of the time $Q_{05}S$ and the flashiness index *FI*.

Discharge measurements used in this studied are monitored by the WPS in the "Aqualim" network (aqualim.environnement.wallonie.be). The Aqualim network is currently constituted of 170 gaging stations recording water depth every 10 minutes. Water depth is measured either with pressure or RADAR sensors. Data are automatically sent through gsm/gprs networks. Data are validated every two weeks through the cross-checking between the recorded water depth and the value read on a staff gauge by an operator. Furthermore, maintenance is organised by the WPS. Water depth-discharge relationships are established and regularly complemented. In Chapter 3, 22 catchments were chosen from this database according to different criteria. The first criterion was that the catchment had to be located in the Ardenne ecoregion mainly in order to control as much as possible the geological factor, as it plays an important role in water fluxes partitioning. Then, chosen catchments have no overlapping area and special characteristics (e.g. presence of a dam). Finally, these choices were done following discussions with the WPS responsible for these measures having intensively used these measurements in his research (Gailliez, 2013) in order for example to avoid gauging stations with known-measurements errors within the studied decade.

2.2.2 Water quality variables

In order to study the impact of forest cover on water quality (see Chapters 4 and 5), we characterized water quality through biological and physico-chemical variables measured as part of the monitoring of water bodies quality performed by the WPS for the EU-WFD (SPW-DGO3-DEE, 2016).

Physico-chemical water quality is described by the following variables: Dissolved Oxygen, Nitrates, Chloride, Sulfates, pH, Temperature, Total Phosphorus, Nitrites, Ammonium, Dissolved Organic Carbon and Suspended Materials. In Chapter 4, a methodology was developed to use as much station data as possible across 10 years, ending up in the study of 362 stations.

Biological water quality is characterized through the macroinvertebrates index and the diatoms index. The macroinvertebrates index is based on the French IGBN (i.e. "Standardized Global Biological Index") adapted to Wallonia (Vanden Bossche, 2005). The IBGN score, with a range from 0 (no indicator taxa) to 20, is obtained by crossing two sub-indices: the "faunal indicator group" reflecting pollution sensitivity and the taxonomic diversity class. The index based on benthic diatoms is the IPS ["Specific Polluosensitivity Index", see Coste in CEMAGREF (1982)]. In Chapter 5, in order to answer more specific questions than these addressed in Chapter 4 (e.g. regarding the effect of the location of forest in the catchment), stations from the EU-WFD monitoring network that monitor headwater waterbodies were selected.

2.2.3 Land use and land cover data

We used the Top10VGIS data set from 2010 from the Belgian National Geographic Institute (NGI, www.ngi.be) to characterize the LULC in this regional study and in particular forest cover and its main different types (i.e. needle-leaved and broadleaved forest). We qualify this dataset and the spatial units studied in our work as LULC, even though these concepts are not interchangeable (Comber et al., 2008). Indeed the dataset itself contains land use classes as 'cropland' and land cover classes as 'forest'.

The TOP10VGIS data set is a vector data set (scale of 1:10 000), which covers the whole of Belgium, is based on the NGI topogeographic data that classifies LULC into 37 classes. The production of this data set in based on photointerpretation of aerial photographs and the completion of information by field operators.

2.3 Statistical methods and software use

2.3.1 Multiple Linear Regression

Regression analysis is a statistical modeling method whose purpose is either to find the best functional model relating a response variable to one (simple regression) or several (multiple regression) explanatory variables, in order to test hypotheses about the model parameters, or to forecast or predict values of the response variable (Legendre and Legendre, 2012a).

In Chapter 3, we used Multiple Linear Regression (MLR) to assess the linear link between five indicators – taken one at a time – characterizing the HES of water supply and water damage mitigation in terms of quantity and timing, and forest cover and climate variables. We considered the necessity to transform variables prior to applying MLR in order to improve normality of distribution and linearity of the multiple relationships between dependent and independent variables.

2.3.2 Principal Component analysis

Principal components analysis (PCA) is a common multivariate method used to summarise, in a low-dimensional space built upon a small number of independent variables, the variance in a multivariate scatter of points (Legendre and Legendre, 2012b). It is an indirect gradient analysis providing an indication of linear relationships between objects and variables of the dataset. It also allows handling data sets with many variables through the collapsing of these many variables into a few independent principal components (PCs), which can be used in further analyses. Olsen et al. (2012) present application of this method to water quality data analysis and Legendre and Legendre (2012b) provide information for deeper understanding. We used this technique in Chapter 3 and 4 to create independent variables representing LULC, and in Chapter 5 to describe biological water quality dataset variability and correlation with supplementary variables. The number of axes that could be interpreted was assessed through the Kaiser-Guttman criterion and we checked if the broken stick model was consistent with that.

2.3.3 Redundancy analysis

Indirect gradient analyses such as PCA are generally applied to describe the structure of a dataset but when it comes to quantify and describe the relationships of two particular sets of variables, direct gradient analysis is more adapted. Among these analyses, redundancy analysis (RDA) is a method combining the properties of ordination and regression methods, used to extract and summarise the variation of a response dataset (containing several variables) that can be explained by a set of explanatory variables. More specifically, RDA allows summarizing linear relationships between components of response variables that are "redundant" with (i.e. "explained" by) a set of explanatory variables. In partial RDA, the linear effects of the explanatory variables on the response variables are adjusted for the effects of the covariables (see Figure 2-2 for fractioning of variation between variables, covariables and residual variation).

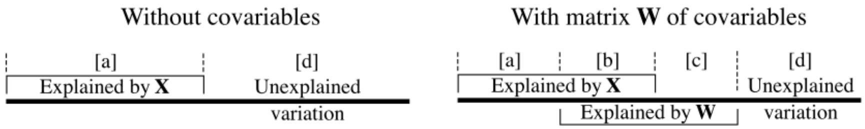


Figure 2-2. Explained and unexplained variation fractions without covariable (left, RDA) and with covariable (right, partial RDA), from (Legendre and Legendre, 2012a)

In classic RDA [see Figure 2-2 (left)], the total variation of the variable of interest (Y) is split into a fraction which is explained by X (i.e. [a]) and a fraction which is not explained by X (i.e. the residual variation, [d]). In partial RDA [see Figure 2-2 (right)], the variation of Y is split into a fraction which is explained by X alone (i.e. [a]), the variation explained by W (i.e. the variation explained by W alone[c] + the variation explained jointly by X and W [b]) and the residual variation [d]. When running both models, [b] and [c] can be derived.

Variation partitioning consists in apportioning the variation of a variable among two or more explanatory data sets. When used in relation to RDA, it is constructed as follows: multiple partial RDAs are run to determine the partial, linear effect of each explanatory matrix on the response data.

We used RDA, partial RDA and variation partitioning in Chapters IV and V to quantify the fractions of variability in water quality – and their significance – explained by forest cover and other environmental variables.

Significance of the RDA models, axes and variables were tested using permutation tests. Their principle is to generate a reference distribution of the chosen statistic under the null hypothesis H0 by randomly permuting appropriate elements of the data a large number of times (in our case 999) and recomputing the statistic each time (Borcard et al., 2011). Then, the true value of the statistic is compared to this reference distribution. The p value is computed as the proportion of the permuted values equal to or larger than the true (unpermuted) value of the statistic for the F test (in RDA). The null hypothesis is rejected if this p value is equal to or smaller than the predefined significance level (Borcard et al., 2011).

2.3.4 Software use

Most data processing was run in the open-source R statistical software (R Core Team, 2013). Several packages were used, we only cite the main ones and in particular, the packages that are not included in the R standard installation. Hydrological indicators (Chapter 3) were computed using integrated packages except for low flows indicators which were extracted from discharges series using the *lfstat* package from Koffler et al. (2015). PCA were computed using *FactoMineR* (Lê et al., 2008) and *vegan* packages (Oksanen et al., 2017). MLR were run in R statistic software with the integrated *stats* package. RDA and variation partitioning were run using the *vegan* package developed by Oksanen et al. (2017).

As a summary, Figure 2-3 presents the materials, methods and study scales for every part of the study.

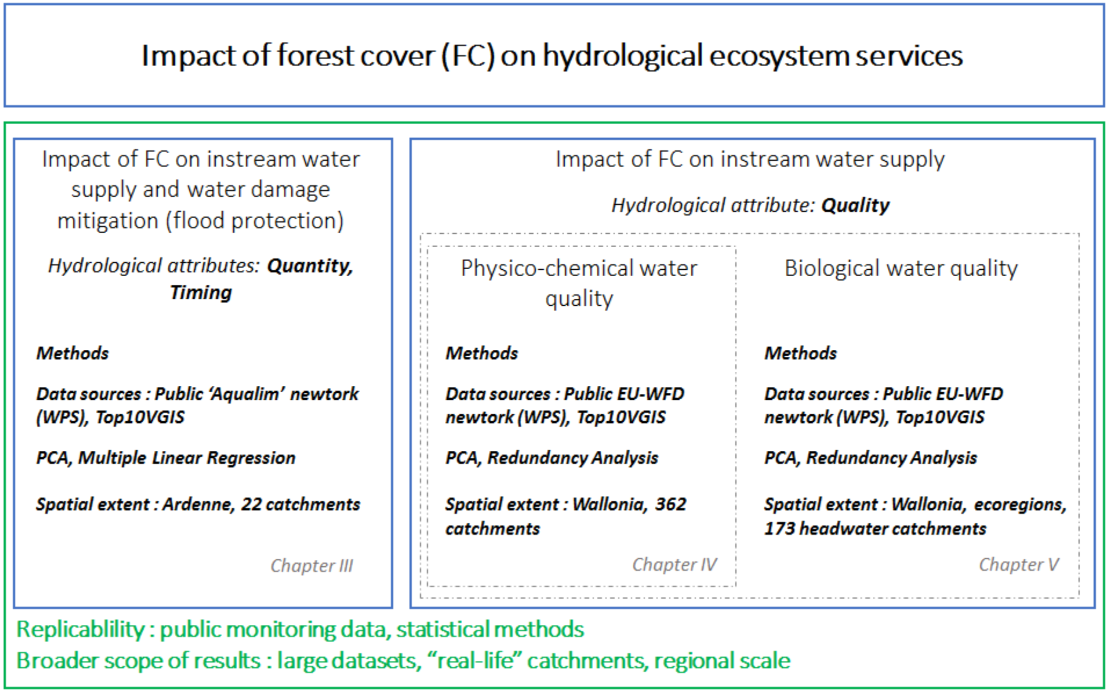


Figure 2-3. Material and methods mobilised to fulfil the main objectives of the study (blue: thematic objectives; green: methodological objectives)

Chapter 3 Forest cover impact on instream water supply and flood protection in terms of quantity and timing

The following text is directly taken from the following published article:

Brogna, D., Vincke, C., Brostaux, Y., Soyeurt, H., Dufrêne, M., Dendoncker, N., 2017b. How does forest cover impact water flows and ecosystem services? Insights from "real-life" catchments in Wallonia (Belgium). Ecol. Indic. 72, 675–685.

Preamble and precisions

This research main objective is the **study of the impact of forest cover on instream water supply and the flood protection service**. In this study, the hydrological attributes of quantity and timing are studied in the Ardenne ecoregion (mainly in order to control as much as possible the geological factor). Water supply and flood protection services are approached through five indicators extracted from 10 hydrological years (2005–2014) discharge data series, as discharge presents the interest of reflecting and integrating the processes partitioning water fluxes and the forest cover effect (see 1.3). These were computed annually and seasonally. The water supply was assessed through the specific volume *Vs*, the baseflow index *BFI* and the specific discharge exceeded 95% of the time $Q_{95}S$. The flood protection service was approached through the specific discharge exceeded 5% of the time $Q_{05}S$ and the flashiness index *FI*. Multiple linear regression (MLR) models were created to assess the forest cover and forest cover type on the studied HES.

Transversal methodological objectives are in here reached, through the development of a replicable approach (i) based on easily accessible data, monitored in many countries, (ii) using robust but simple and straightforward statistical methods and (iii) with main processes run in open source statistical software. The enlargement of scope of the derived results aiming to come up with land planning recommendations is reached through the study of ES at "real-life" catchments scale ranging from 30 to 250 km^2 with mixed land covers with a focus on forest cover.

Precisions:

The terms "growing period" and "non-growing period" would better fit in the text than the terms "vegetation period" and "non-vegetation period" respectively. However, as the article has been published using these latter terms, we prefer to leave the article as is.

In Figure 3-6, every dot represent a value of monthly rainfall for a specific catchment.

Abstract

While planetary boundaries are being crossed and ecosystems degraded, the Ecosystem Services (ES) concept represents a potential decision-making tool for improved natural resources management. The main aim of this paper is to assess the impact of forest cover on water related ES in Wallonia (Belgium) in terms of quantity and timing. We developed an approach based on easily accessible data, monitored in several countries and using straightforward statistical methods. This led us to study ES at "real-life" catchments scale: 22 catchments – from 30 to 250 km^2 – with mixed land covers were studied. We approached the water supply and flood protection services through 5 indicators extracted from 10 hydrological years (2005–2014) discharge data series. These were computed annually and seasonally (vegetation period from March to September and "non-vegetation" period the rest of the year). The water supply was assessed through the specific volume Vs, the baseflow index BFI and the specific discharge exceeded 95% of the time Q_{95S}. The flood protection service was approached through the specific discharge exceeded 5% of the time Q_{05S} and the flashiness index FI. Our study gives two main insights. First, statistical analyses show that forest cover negatively impact water supply when studying annual and "non-vegetation" flows in general (Vs) but positively when studying low flows (Q_{95S}). Regarding flood protection the study did not show any significant effect of forest on annual high flows (Q_{05S}) but a negative impact in the "non-vegetation" period. Forest cover showed a negative impact annually and in the vegetation period on the

flashy behaviour of the catchment thus a positive effect on the flood protection ES. The “year” effect is overall highly significant, testifying the importance of climatic factors. Rainfall is often significant and can be considered as a main driver of these ES. Secondly, the quality of the models produced and the results overall we assume – in line with the literature – that other variables characterizing the catchments such as topography or soil types do play a significant role in these ES delivery. This questions the use of land cover proxies in assessing and mapping of hydrological ES at a complex landscape scale. We thus recommend further research to keep improving land cover proxies when they are used.

3.1 Introduction

Ecosystem Services (ES) can be defined as the benefit people obtain from nature (Millennium Ecosystem Assessment, 2003). In the present context of the overtaking of planet boundaries (Rockström et al., 2009; Steffen et al., 2015) and the degradation of ecosystems and their services (Costanza et al., 2014; Millennium Ecosystem Assessment, 2005b) the ES concept can raise awareness about the importance of preserving ecosystems and biodiversity (Millennium Ecosystem Assessment, 2005b). Haines-Young and Potschin (2010a) suggest a representation of this concept at the interface of ecosystems and human well-being in the form of a 'cascade' model. This framework is based on the idea that a sort of 'production chain' starts from the ecosystems biophysical structures and processes which lead to functions that create services providing benefits and socio-economic values to human beings. Human society retroacts on ecosystems through pressures but also restoration actions. Following this view, the ES concept has the potential to be a decision-making tool for improved natural resources management (de Groot et al., 2010). To achieve this potential there is a need for quantifying accurately every component of the ES 'cascade' through suitable indicators (Braat and de Groot, 2012; Müller and Burkhard, 2012; Seppelt et al., 2011). In order to do so, one of the main challenges is to deal with the high complexity of the ecosystems functioning and the complex dynamics characterizing the links between processes – functions and services at different temporal and spatial scales (Bastian et al., 2012; Carvalho-Santos et al., 2014; de Groot et al., 2010; Swetnam et al., 2011; Turner and Daily, 2008; Villa et al., 2014). Assessing ES to support land planning decision-making remains thus a challenge due to multiple sources of uncertainty such as data scarcity, functional knowledge gaps, demand variability, etc. (Jacobs et al., 2013). In practice however, and even if research is done to improve them, land cover based proxies are used in local or national ES assessments (Albert et al., 2015; Koschke et al., 2012; UK NEA, 2014). Arguably, policies such as the EU biodiversity strategy 2020 targets (European Commission, 2011) requiring member states to assess and map ecosystems and their services provide an incentive for using such techniques. Indeed, these methods, and in particular one commonly used known as the 'matrix model' (Burkhard et al., 2010), allow for straightforward ES mapping. However, one may question the validity of these maps as uncertainties are high and variable, in particular in terms of the expected direct and univocal links between land cover and ES provided.

Among ES being mapped, those related to water are of prime importance. Water is indeed the most essential component for the life of all beings, it is at the core of sustainable development and is of major importance to healthy ecosystems, socio-economic development and to the survival of human beings (Haddadin, 2001; UN-Water, 2014). Land cover and in particular forests have an impact on hydrological services through their impact on water cycle flows [see figure 3-1 for an adaptation of the ES 'cascade' model to hydrological services provision by forests by Carvallos-Santos et al. (2014)]. As shown on figure 3-1, these services are delivered according to three dimensions: quantity (i.e. total water yield), timing (i.e. seasonal distribution of flows) and quality (i.e. removal and breakdown of pollutants and trapping of sediments) (Brauman et al., 2007). Forests are seen as the main ecosystems interacting with water, due to their height, dense and irregular crown canopy resulting in a high leaf area index and lower albedo, architecture of their spread root system widely prospecting soil horizons, wide horizontal distribution and vertical coverage (Calder, 2002; Salemi et al., 2012; Zhang et al., 2001). By explicitly listing operating processes and functions, figure 3-1 highlights the complexity of water-related ES assessments as different functions may impact the same hydrological service in opposite ways.

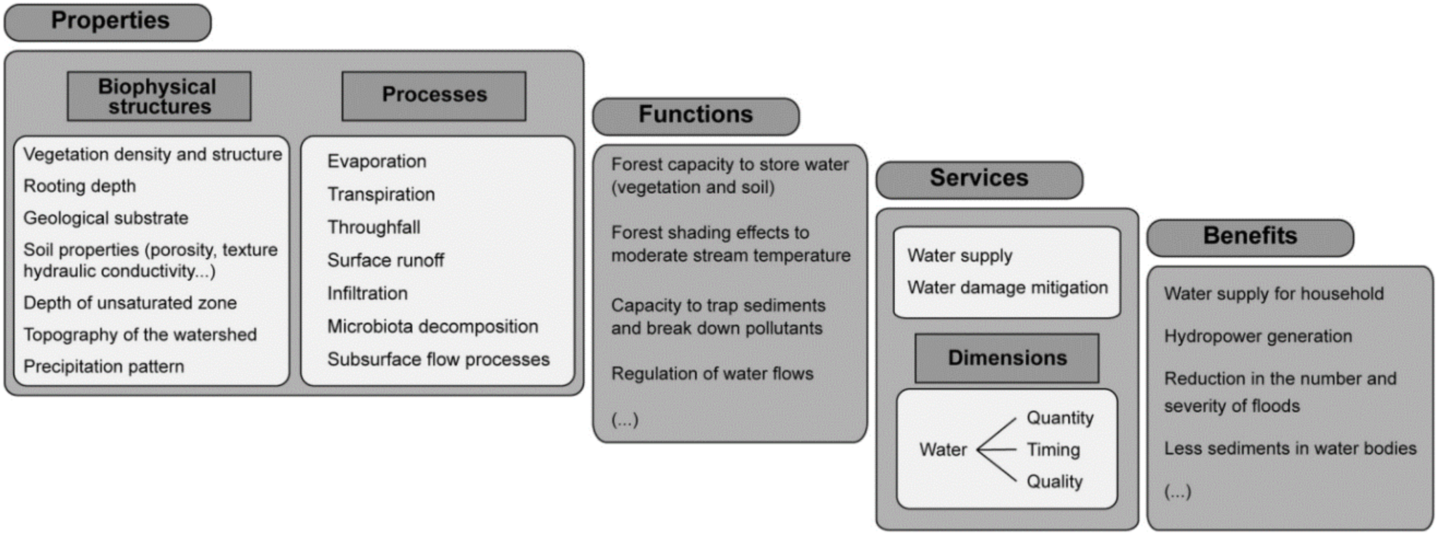


Figure 3-1. Conceptual framework for hydrological services provision by forests, from Carvalho-Santos et al. (2014).

Despite the fact that many studies were conducted to measure the impact of forest cover on water cycle components (Brown et al., 2005; Farley et al., 2005; Robinson et al., 2003), relationships between water flows (quantity and timing) and forests have been controversial since Pliny the Elder (Andréassian, 2004b). Nevertheless, the associated assumed effects on processes are a high evapotranspiration, the promotion of infiltration compared to surface runoff or rapid drainage (at least on low slopes), increase of soil moisture content, recharge of groundwater and the gradual release of water (Aussenac, 1996; Bruijnzeel, 2004; Calder, 2002; Office National des Forêts, 1999). Many of these studies are paired-catchment studies [see Bosch and Hewlett (1982) for a review] where catchment size is for the vast majority less than 2 km^2 limited by the fact that these experiments require controlling most of the factors impacting water flows while having pure and distinct land covers between catchments. These studies report an increase of annual water yield when forest cover is replaced by lower vegetation cover. At a global scale, which is out of the scope of this study, authors argue that forest cover raises the precipitation events likelihood and increases water yield by contributing to the availability of atmospheric moisture vapor and the transport across continents (see Ellison et al. (2012) for a “forest-water yield” debate review). In this context of inextricable link between forest and water, many authors acknowledge the fact that more research is needed to study the impact of forest cover on water fluxes at different latitudes, in different contexts (e.g. different soil types) and at different spatial and temporal scales (Brown et al., 2013; Cosandey et al., 2005; Garmendia et al., 2012; Price, 2011). Regarding hydrological services assessment and the impact of land cover on these ES, the ‘catchment’ appears to be a relevant spatial unit of study because of its integrative character (Granier, 2007) and its reality as component of the landscape. Numerous studies focus on measuring precisely the water cycle fluxes at the stand scale but can hardly be extrapolated (Oishi et al., 2008; Schume et al., 2003; Schwärzel et al., 2009; Unsworth et al., 2004; Vincke et al., 2005; Wilson et al., 2001). Carvalho-Santos et al. (2015) assess and map hydrological services at the catchment scale through physically based modelling, highlighting the fact that daily rainfall – runoff models were stated to be really robust methods by Crossman et al. (2013) but are not often used in the ES sphere. Indeed these require vast amount of data, are complex and time consuming to calibrate and are often applied on one single catchment. Finally global changes push scientists to claim for renewing studies linking hydrological processes and land cover. Global changes which affect water quality and quantity (climate change, land use change and invasive species) question the assumption that studies from the last decades can be used to face future conditions (Bates et al., 2008; Huntington, 2010; Vose et al., 2011).

Regarding this context, the main aim of this paper is to assess the impact of forest cover on water related ES (i.e. water supply and water damage mitigation) in Wallonia (Belgium) in terms of quantity (i.e. water yield) and timing (i.e. seasonal distribution of flows). In order to do so and to ensure replicability, we developed an approach (i) based on easily accessible data, monitored in many countries, (ii) using robust but simple and straightforward statistical methods and (iii) with main processes run in open source statistical software. This will lead us to study ES at "real-life" catchments scale ranging from 30 to 250 km² with mixed land covers with a focus on forest cover. In doing so, we also aim to provide information to the debate of using land cover proxies versus more advanced methodologies to derive indicators used to map water related services at a complex landscape scale but meaningful in land planning processes.

This document is structured as follow: first we present the study area in Section 3.2; then we describe our approach globally and then detailing the hydrological (i.e. extraction of hydrological indicators) and physical description of the studied catchments, the rainfall over the period of study computation, and the study of the impact of forest cover on hydrological services in itself. Third, we present the results and finally discuss them in section 3.5, highlighting key findings but also discussing strengths and limitations of our study and presenting research perspectives.

3.2 Study Area

The study area corresponds to the "Ardenne" region (4°7'42'' to 6°24'40'' E, 42°27'00'' to 50°41'00''N in WGS84 geographic coordinate system; Figure 3-2). The Ardenne is an ecologically, geologically and lithologically relatively homogeneous region located to the South-East of Wallonia (South of Belgium). It covers 5711 km² corresponding to 33% of the Walloon region and 19 % of Belgium. This high plateau dissected by several rivers constitutes the western protruding end of the "Rhine great schistose massif" (see Noirfalise (1988) for a more detailed description). This region was chosen for several reasons. The main one is that focusing on this study area allows to best control the geological factor, which plays an important role in the spatial distribution of the water (Grandry et al., 2013). The Ardenne is entirely located on the same aquifer: "the Primary schistose and sandstone massif". This region also presents relatively homogeneous climatic and pedological characteristics (De Slover and Dufrêne, 1998). Finally it contains nearly two thirds of the Walloon forests, focus land cover of this study. The forests of Ardenne are composed at 15% of needle-leaved forests, 9% of broad-leaved forests and 6% of mixed forests. The broad-leaved forests are generally natural to semi-natural. The species composition is mostly dominated by oaks [*Quercus robur* L. and *Quercus petraea* (Matt.) Liebl.] and beech (*Fagus sylvatica L.*) stands, but other species such as birch (*Betula pubescens* Ehrh. and *Betula pendula Roth.*), ash (*Fraxinus excelsior* L.), maple (*Acer pseudoplatanus* L.), and hornbeam (*Carpinus betulus* L.) are also common. Needle-leaved forests can be considered as an artificial forest type regarding the species composition – needle-leaved tree species forests are not native in Belgian forests – but also the common occurrence of drainage infrastructure. Most of the time (> 80%), needle-leaved forests are composed of even-aged stands of Spruce (*Picea abies* (L.) H. Karst). Douglas fir (*Pseudotsuga menziesii* Mirb. Franco), larches (*Larix sp.*), and pines (*Pinus sylvestris* L. and *Pinus nigra* R. Legay) are also regularly present (Alderweireld et al., 2015).

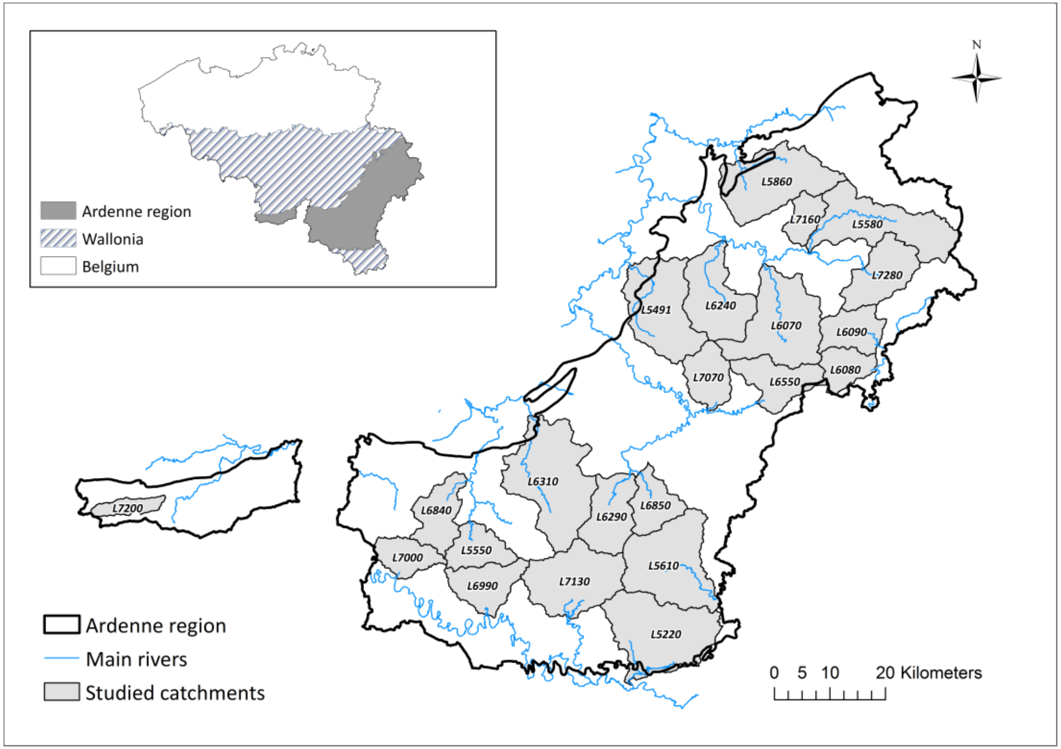


Figure 3-2. Studied catchments location in the Ardenne (Wallonia, Belgium)

We selected 22 independent catchments (i.e. with no overlapping areas; see figure 3-2) from the Walloon Public Service (WPS) “Aqualim” hydrological monitoring network across this region (aqualim.environnement.wallonie.be). This catchment scale approach was preferred to studies at finer scales to try to encompass the complexity of the processes occurring in the water cycle. This unit of study is particularly interesting because of its integrative character (Granier, 2007) and therefore makes sense in an assessment of the impact of forest cover on ecosystem services at the landscape scale.

3.3 Material and Methods

The developed methodology to assess forest cover impact on hydrological ES comprises 5 main steps (see figure 3-3): 1) extraction from the discharge data series of synthetic indicators (referred to as “Yi” variables in the multiple regressions – see step 5) characterizing the hydrological behaviour of the studied catchments in relation with water supply and water damage mitigation ES (in this study, flood protection service); 2) definition of physical characteristics of the catchment from land cover, topographic and soil datasets. These are either the focus dependent variable, i.e. land cover or catchment description variables; 3) application of Principal Component Analysis (PCA) on land cover classes and extraction of sites scores for the first 2 principal components (PC1 and PC2); 4) computation of daily rainfall for each catchment for the whole period as a control factor; and 5) multiple linear regressions of each “Yi” towards the year factor, rainfall and the obtained PCs.

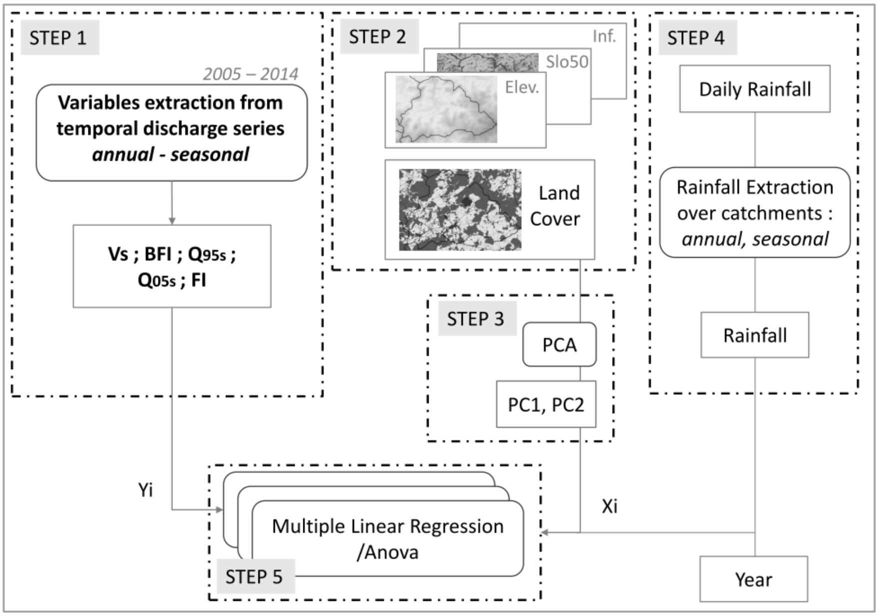


Figure 3-3. Methodology framework to study the impact of land cover and particularly forest cover on water supply and water damage mitigation (in this study, flood protection). Extraction of hydrological indicators (Yi, with Vs: Specific Volume, BFI: baseflow index, $Q_{95}s$ and $Q_{05}s$: specific discharge exceeded 95 and 5% of time, FI: flashiness Index) which are regressed against rainfall, year effect and the first two P(Yi, with Vs: Specific Volume, BFI: baseflow index, $Q_{95}s$ and $Q_{05}s$: specific discharge exceeded 95 and 5% of time, FI: flashiness Index) rincipal Components (PC1, PC2) of land cover Principal Component Analysis (PCA).

3.3.1 Hydrological and physical catchments description

3.3.1.1 Hydrological behaviour of the catchments in relation to ecosystem services supply

The water balance approach is often used when studying the role of a forest cover type on the water cycle [see Office National des Forêts (1999) for an application in the forest context].

This method is based on the principle of continuity and the water balance of a catchment can be written as such:

$dS/dT = P - Q - AET$ [Equation 3.1]

with S : catchment storage (m^3), P : Precipitation (m^3/s), Q : discharge the catchment outlet (m^3/s), AET : Actual Evapotranspiration (m^3/s).

Discharge at the catchment outlet is thus reflecting and integrating the processes partitioning water fluxes. Furthermore, it is a variable that is widely and with a high frequency monitored. We extracted indicators of hydrological behaviour of the studied catchments based on daily discharge data for 10 hydrological years (October 2004 to September 2014) provided by the WPS (aqualim.environnement.wallonie.be). We chose the hydrological year as the annual base period, temporally defined by the period between October and September of the following civil year (e.g. in this paper the hydrological year 2005 starts on the 1st of October 2004 and ends on the 30st of September 2005).

Table 3-1 lists the indicators computed to characterize the hydrological behaviour of the catchments and the associated ES. These indicators were selected to cover different aspects of the hydrological regime (i.e. global regime, low and high flows) and to be linked to water supply and flood protection ES. We processed data of ten hydrological years (i.e. from 2005 to 2014) for two reasons: (i) these are simultaneous to the collection of data for the creation of the land cover map and (ii) covering several years allows us to cover different rainfall amounts. We computed these indicators annually and seasonally. We divided the year into two seasons to reflect the phenological variability of the vegetation: firstly, the “vegetation season” from April to September and secondly the “non-vegetation” period from October to March.

Table 3-1. Indicators of the hydrological behaviour of catchments and related ecosystem services

Hydrological indicator	abbreviation	Ecosystem service
Hydrological regime		
Specific volume	*Vs*	water supply
Flashiness Index	*FI*	<> flood protection
Low flows		
Base Flow Index	*BFI*	water supply
Specific 95th discharge quantile	$Q_{95}s$	water supply (low water context)
High flows		
Specific 5th discharge quantile	$Q_{05}s$	<> flood protection

Daily discharges data were aggregated to annual and seasonal specific volumes (Vs, see table 3-1 and Eq. (1)). Specific volume is an indicator of the water supply ES (Carvalho-Santos et al., 2015; Garmendia et al., 2012). The specific volume for a period is defined as the total streamflow for that period divided by the catchment area:

$$Vs = \frac{\sum_{d=1}^{N} Q_d * 86\,400}{S} \;(m^3/m^2) \qquad \text{[Equation 3.2]}$$

where Q_d represents the daily discharge (m^3/s), N the number of days for that period and S the catchment area (m^2).

To characterize low flows and the water supply service from another point of view, the baseflow index (BFI) was computed seasonally and annually. The BFI is the proportion of baseflow to total streamflow over a continuous period of record (Bloomfield et al., 2009). BFI represents the way the soil infiltrates water and returns it to the stream. Computing this index requires the separation of baseflow from stormflow. In this study the BFI was computed using the *lfstat* package (see Gustard and Demuth (2008) for complete description of the methods) of the R statistical software (Koffler et al., 2015).Using the same package we also computed the annual and seasonal specific 95th quantiles (Q_{95}s, defined as the discharge exceeded 5% of the time divided by the catchment area) as indicator of low flows (Braud et al., 2013) that can be related to water supply for human consumption but also for riparian and aquatic habitat provision.

We computed the specific 5th quantile ($Q_{05}s$ defined as the discharge exceeded 5% of the time divided by the catchment area) annually and seasonally as an indicator of high flows (Braud et al., 2013; Carvalho-Santos et al., 2015). The flashiness of the hydrological regime was assessed through the computation of the flashiness index (FI) as the ratio Q_{05}/Q_{95} (Jordan et al., 2005). FI and $Q_{05}s$ are indicators that can be inversely linked to the flood protection service.

3.3.1.2 Physical characteristics of the catchments

We used the TOP10VGIS land cover data set provided by the Belgian National Geographic Institute (NGI, www.ngi.be) to characterize the studied catchments' land cover. This vector data set covering the whole of Belgium contains the NGI topogeographic data describing the land cover in 37 classes. For the purpose of the present study the original land cover classes were either selected as such or aggregated ending up finally with seven classes of interest: needle-leaved forests, broad-leaved forests, mixed forests, crops, grassland, water surfaces, shrubs - heathlands and moorlands. We computed percentages of these classes in the studied catchments. Because of the lack of data describing land cover on a yearly basis and based on the same methodology, we assumed that the evolution of the retained classes was minor throughout the studied decade. This was checked through Corine Land Cover 1990, 2000 and 2006 datasets comparisons (http://www.eea.europa.eu/publications/COR0-landcover). Analysis show that change in level 1 classes was always under 3%.

We computed average elevation and median slope over the catchments as physical descriptors from the ERRUISSOL digital terrain model provided by the WPS (http://geoportail.wallonie.be). Regarding the soil infiltration capacity we used an 'infiltration map' covering Wallonia provided by the WPS. This map is based on the Walloon numerical soil map. It takes into account soil texture, drainage characteristics, substratum and, when appropriate, percentage of stoniness and aims at reflecting soil infiltration capacity (Demarcin et al., 2011). Soils are classified into 5 classes representing categories of the soil infiltration capacity in millimetre per hour (mm/h): class 1: Unclassified, superior limits of the other classes are 10.2 mm/h for class 2, 7.6 mm/h for class 3, 3.8 mm/h for class 4 and 1.3 mm/h for class 5. We computed the percentage of the catchment of each 'infiltration map class' as soil descriptor (IC2 to IC5) in order to feed – along with average elevation and median slope of the catchment – the analysis and discussion of our study.

We run principal components analysis (PCA) on the catchments' land cover classes (except one to ensure non collinearity of the variables). This PCA was run on non-standardized variables as they are homogeneous in terms of units (i.e. %) and as we aimed at highlighting main trends and keep the effect of main land cover classes. Based on the Kaiser-Guttman criterion [see Guttman (1954); Legendre and Legendre (1998)] we selected the first two principal components (PC) to use them as uncorrelated explanatory variables in the multiple linear regressions. We added descriptive variables on the PCA plot to describe the catchments.

3.3.2 Rainfall description

Daily rainfall data for the period of interest were provided by the Royal Meteorological Institute of Belgium (RMIB). Journée et al. (2015) provide information on the rain gauge network and methods of interpolation used to create rainfall maps. The data used in this study were provided on a 5 by 5 km grid. The grid was created by kriging interpolation – ordinary or external drift kriging depending on correlation with topography (Wackernagel, 1996). The relatively low spatial resolution of this grid does not seems to be inappropriate given the high spatial auto-correlation of rainfall data (i.e. values from one pixel will be close to values of adjacent pixels, and intra-pixel variability is expected to be rather small given the relatively homogeneous properties of pixels in terms of altitude for example) and given the relatively large study extent (i.e. 22 catchments spread over more than 5000km^2). We computed daily rainfall over the studied period based on the catchments boundaries and the rainfall grid data. This was done through averaging daily rainfall amount of grid centroids contained in the catchment and a buffer zone of 1250m around the catchment. Monthly, seasonal, and annual rainfall were computed based on these daily rainfall amounts over the catchment. We used these values to describe the overall rainfall regime of the 10 hydrological years of study and to control for the rainfall factor in the multiple regressions.

3.3.3 Effect of land cover on hydrological indicators

To ensure replicability and cover a wide range of “real-life” catchments we chose to apply multiple linear regression – a common and simple statistical method – in order to study the impact of land cover on hydrological services. After examination of the variables’ distributions, we applied log-transformation on some explanatory and dependent variables to improve normality of distribution and linearity of the multiple relationships between Yi and Xi. Automatic procedures were set up to perform multiple linear regressions aiming at trends detection over 10 years.

The regression model tested is:

$$Y_i = f(Year + Rainfall + PC1 + PC2) \qquad \text{[Equation 3.3]}$$

with Y_i : hydrological indicator, Year : categorical factor, Rainfall : rainfall amount (mm) during the period over the catchments, PC1 and PC2 : coordinates of the catchments on the first and second PCs respectively of the land cover PCA.

We chose to first include the “year” effect and the rainfall variable in order to correct for the inter-annual climate variability and in particular the amount of rain which is one of the main drivers of the system. This was preferred to a simple division of the hydrological indicator (such as Volume) by rainfall amount to avoid misassumption about the relationship between the indicators and rainfall.

3.4 Results

3.4.1 Catchments description

The 22 catchments’ areas and elevation range from 31 to 247km^2 and 290 and 558m respectively. The main land cover types are forests and grasslands (see boxplot in Figure 3-4). Within the forest class which covers between 26 and 71 % of the studied catchments, needle-leaved forests are more represented than broad-leaved and mixed forests. Figure 3-4 illustrates the heterogeneity of land cover within catchments: few catchments have more than 50% of the same land cover. Needle-leaved forests cover between 8 and

53% of the catchments while broad-leaved forests are less represented with a minimum and maximum cover of 3 and 33% respectively. Grassland cover ranges from 19 to 64%. Artificial surfaces, shrubs - heathlands and moorlands, and mixed forest are hardly represented with low variability.

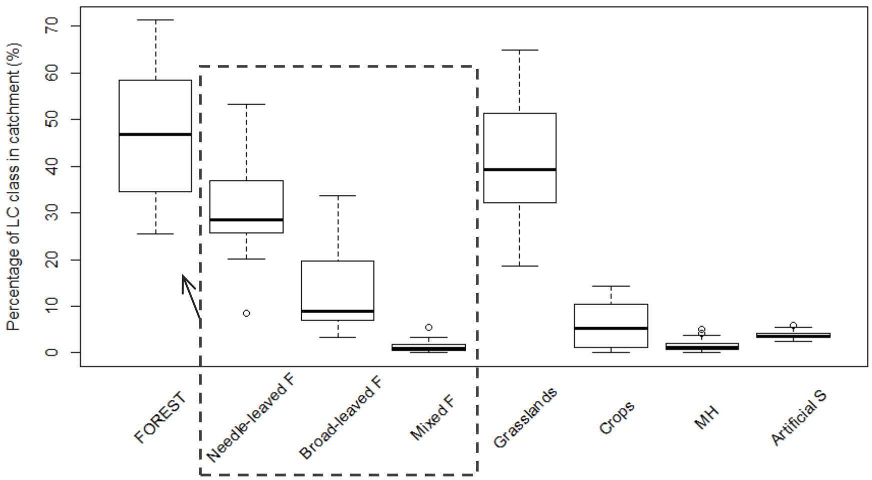


Figure 3-4. Boxplot of land cover types percentages in the 22 studied catchments with F: Forest, MH: shrubs - heathlands and moorlands and S: Surfaces

The first two PCs of the unscaled PCA conducted over the land cover percentages explain 64% and 29% of the dataset variability respectively (see PCA biplot and individuals factor map in Figure 3-5.

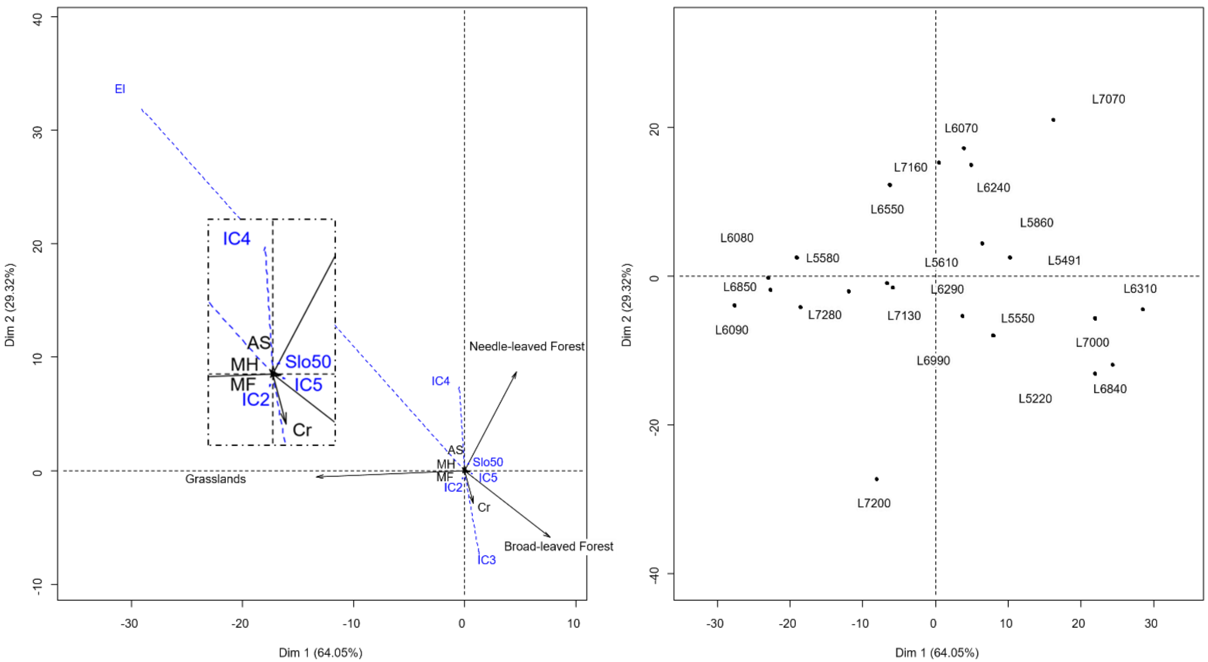
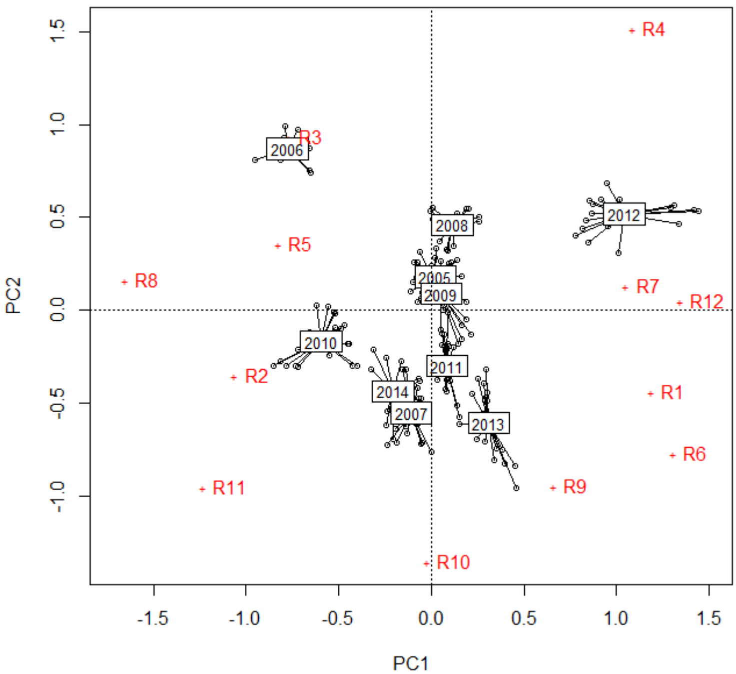


Figure 3-5. Principal component analysis on the land covers percentages over the 22 catchments. Left: Biplot, with AS: artificial surfaces, MF: mixed forest, MH: shrubs - heathlands and moorlands, Cr: crops. Infiltration classes percentages over the catchments (from good (IC2) to bad infiltration rate (IC5)), elevation (El) and median slope (Slo50) variables were drawn on the land cover PCA space. Right : Individuals (catchments) factor map.

3.4.2 Rainfall over the studied period

We computed daily seasonal and annual rainfall. In line with the rest of the study, we worked with hydrological years. The first 3 PCs of the PCA run on monthly rainfall (10 year) over the 22 catchments explain 29, 16 and 14 percent of the variability of the dataset respectively. Biplots based on these three PCs illustrate that the inter-annual variability of rainfall is higher than the variability between catchments for the same year (Figure 3-6). This enforces the choice to treat each year separately and not aggregate indicators over several years.

A



B

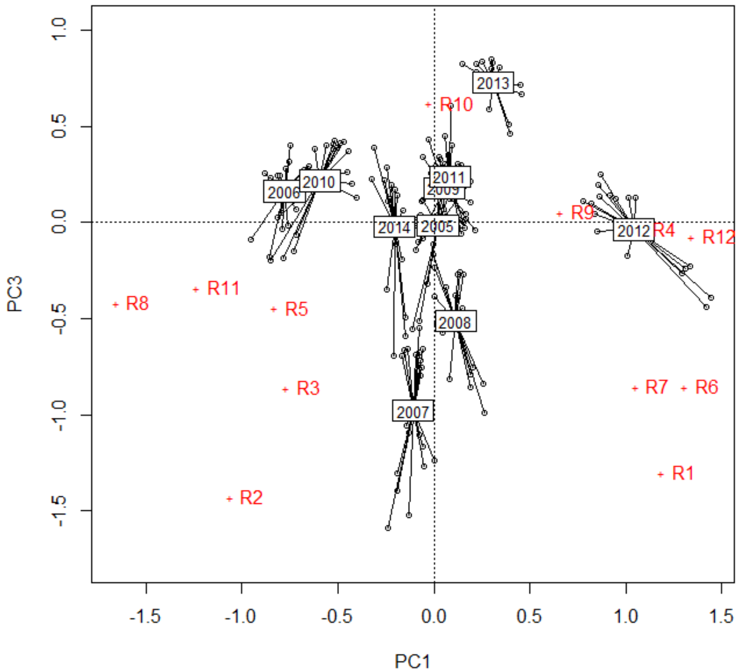


Figure 3-6. Monthly rainfall PCA biplot (A: PC2 vs PC1, B: PC3 vs PC1) with year labelling, cross show the end of variables arrows (R1 = January rainfall → R12: December rainfall)

On this basis we describe annual rainfall characteristics over the studied period. Boxplots in figure 3-7 and figure 3-8 shows the distribution of annual and seasonal rainfall across catchments respectively. Regarding the "normal values" of annual rainfall (1981 – 2010) ranging for the Ardenne region from 900 to around 1400 mm, we can consider the studied years as representative of the region. 2011 was the driest year overall with a really dry vegetation period whereas 2007 and 2008 were the rainiest years annually and seasonally.

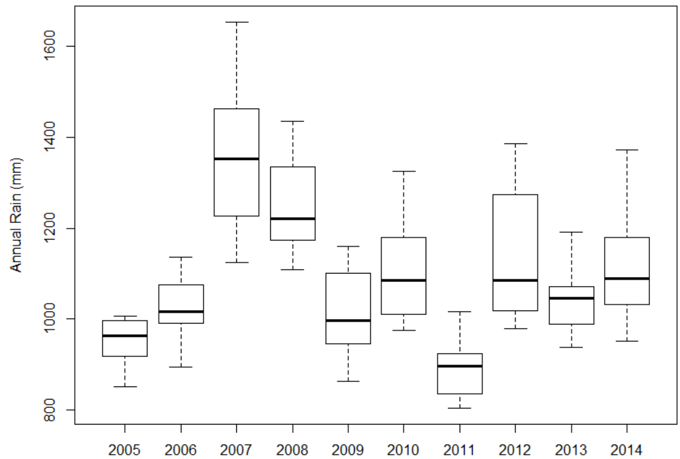
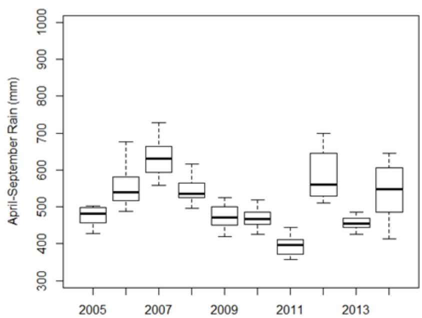


Figure 3-7. Boxplot of annual rainfall (mm) over the 22 catchments of study.



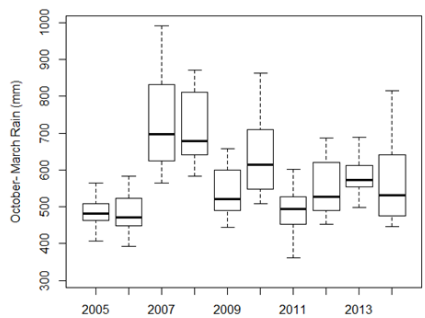


Figure 3-8. Boxplots of seasonal rainfall (mm) over the 22 catchments of study (left: vegetation period; right: "non-vegetation" period)

3.4.3 Effect of land cover on hydrological indicators

Results of the multiple linear regressions on annual and seasonal hydrological indicators during the studied period (2005 – 2014) are presented in Table 3-2. Every indicator except the BFI was log-transformed as the Rainfall variable to improve normality of distributions and linearity of the multiple relationships between Yi and Xi. For every indicator, models are presented for different temporal period (in columns): (i) annually, (ii) seasonally: April to September and October to March. Figures in the table indicate the ratio of the beta coefficient (i.e. the figure multiplying the explanatory variable in the regression model) to the standard deviation of that variable; the degree of significance of the variable is shown on the right. As expected, the year effect is highly significant for every indicator irrespective of the defined temporal period. This finding confirms the interest to include this variable in the used models as well as the interest to cover several sampling years in this study.

*Table 3-2. Results of multiple linear regressions on annual and seasonal hydrological indicators during the studied period (2005 – 2014). R squared of the model (R^2), the figures in the table indicates for each indicator type and for each period, the ratio of the beta coefficient multiplying the variable (see names in first column) in the regression model to the standard deviation of the variable in the dataset (the stars on the right to the figures represent the significance degree of the variable in the model with p values [0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1]), with ‘NL’ = needle-leaved.*

ES	Hydrological indicators (regression components)	Annual		Veg season (April - Sept.)		Non veg season (Oct. - March)	
water supply	Vs						
	R^2	*0.68*		*0.83*		*0.72*	
	Year		***		***		***
	Rainfall	24.9230	***	35.3386	***	14.3954	***
	PC 1 (+ Forest)	-0.0028	**	0.0019		-0.0041	***
	PC2 (+ NL Forest)	0.0019	.	0.0032	.	0.0025	*
	BFI						
	R^2	*0.22*		*0.36*		*0.27*	
	Year		***		***		***
	Rainfall	-4.8680	.	-16.2403	***	-2.0943	
	PC 1 (+ Forest)	-0.0009		0.0015		-0.0018	
	PC2 (+ NL Forest)	0.0028		0.0018		0.0033	.
	$Q_{95}s$						
	R^2	*0.46*		*0.49*		*0.67*	
	Year		***		***		***
	Rainfall	25.9805	***	23.6996	***	6.8500	*
	PC 1 (+ Forest)	0.0063	*	0.0070	*	0.0014	
	PC2 (+ NL Forest)	0.0139	***	0.0106	**	0.0140	***

ES	Hydrological indicators (regression components)	Annual		Veg season (April - Sept.)		Non veg season (Oct. - March)	
<> flood protection	$Q_{05}s$						
	R^2	*0.59*		*0.82*		*0.67*	
	Year		***		***		***
	Rainfall	24.0842	***	40.4988	***	13.5733	***
	PC 1 (+ Forest)	-0.0013		0.0009		-0.0024	.
	PC2 (+ NL Forest)	-0.0014		-0.0001		0.0001	
	FI						
	R^2	*0.36*		*0.46*		*0.68*	
	Year		***		***		***
	Rainfall	-1.8963		16.7992	*	6.7233	**
	PC 1 (+ Forest)	-0.0076	*	-0.0061	.	-0.0038	
	PC2 (+ NL Forest)	-0.0153	***	-0.0107	**	-0.0139	***

3.4.3.1 Specific volume

R^2 coefficients range from 0.68 for the annual indicator to max 0.83 for the vegetation period and 0.72 for the non-vegetation period. The rainfall variable is highly significant and has a positive influence on the specific volume. The “forest versus grasslands PC” (PC1) has a significant negative effect on the variable for the annual and non-vegetation periods. The second PC (PC2) has a significant positive effect for all periods.

3.4.3.2 Baseflow index

R^2 range from 0.22 for the annual indicator to 0.36 for the vegetation period. Rainfall has a significant effect on the BFI except for the “non-vegetation” period. The models do not reveal a statistical effect of the “forest versus grassland” PC (PC1). PC2 influences slightly positively BFI during the non-vegetation period.

3.4.3.3 Specific Q_{95}

R^2 range from 0.46 for the annual indicator to 0.67 for the non-vegetation period. Rainfall is significant in every model with a positive influence on the studied indicator. The forest PC (PC1) has a significant positive effect on the specific 95 discharge annually and during the vegetation period. PC2 also has a significant positive effect for every model.

3.4.3.4 Specific Q_{05}

The 95th percentile of the specific discharge (i.e. Q_{05}s) was modelled with R^2 ranging from 0.59 for the annual indicator to 0.82 for the vegetation period. Rainfall is highly significant in every model with a positive influence on the indicator. The forest PC (PC1) has a slightly significant negative effect during the non-vegetation period overall.

3.4.3.5 Flashiness Index

R^2 coefficients range from 0.36 for the annual indicator to 0.68 for the vegetation period. The rainfall variable is significant with a positive influence on the flashiness for the seasonal periods and none for the annual index. The 'forest versus grasslands PC' (PC1) has a significant negative effect on the variable for all the periods.

For each indicator we observe an improvement of the model quality when subdividing the year into seasons, reflecting the intra annual variability of the rainfall and land cover effects.

3.5 Discussion

3.5.1 Influence of forest cover on hydrological services

3.5.1.1 Preamble: forest cover in our “real-life” catchments

Forest cover is mainly represented in our study through the first component (PC1, Figure 3-5) of the PCA conducted over land cover classes of our studied catchments. To some extent, the type of forest can be related to PC2 which is related to the naturalness of the forest. This technique allowed us to decorrelate variables that were highly correlated (percentages of LC classes) and thus satisfy the variables independence assumption of multiple linear regression method. PC1 explains 64% of the variability of the dataset and clearly opposes catchments with high percentage of grassland cover to catchments with high percentage of forest cover. When we state “forest cover” is in this study, we are thus referencing “real-life” catchments with mixed land covers but high forest cover percentage. Another important aspect is that land cover is also often linked to other factors like soil type and topography (Figure 3-5); still, as discussed below, regression models give us insight into the impact of forest in the Belgian Ardenne in a regional context.

3.5.1.2 Water supply

We approached the water supply service though three hydrological regime indicators: the specific volume, the baseflow index and the specific discharge exceeded 95% of time. A significant part of the specific volume can be described by our model taking into account rainfall, year effect and land cover PCs (R^2 of around 0.70). According to our findings, the effect of forest on annual specific volume is negative in our study area. This negative effect is in line with numerous studies that observed through paired-catchments designs an increase in annual water yield when vegetation such as grasslands are implemented in place of forests or a decrease in annual water yield when afforestation is operated (Bosch and Hewlett, 1982; Brown et al., 2005). We believe this impact is also partially explained by the location of the catchments. In our dataset the catchments with low proportion of forest are located on higher zones being classified by Van der Perre et al. (2015) into the “Cold Ardenne” bioclimatic class. On the other extreme of the PC1 (Figure

3-5), catchments are located in the other bioclimatic class of the Ardenne being defined as the “Hot Ardenne” (Van der Perre et al., 2015). We assume that temperature is also part of and reinforces this effect of forest reducing specific volume through rising evapotranspiration rates. When looking at the seasonal models this negative effect of forest cover is observed in the “non-vegetation” period (October to March) but is not significant during the vegetation period. There is a significant positive influence of PC2 on specific volume. As a reminder the second PC opposes catchments with high needle-leaved forests proportion located on higher elevations and on soils with lower infiltration capacity and steeper slopes to broad-leaved forests (lowest values of PC2) – in association with high percentage of needle-leaved forests – and crops on soils with better infiltration capacity to a lesser extent. In this context we assume that local conditions (soil types, soil conditions and topography) have a major impact on specific volume (soils with low infiltration capacity and on steeper slopes being correlated with higher specific volume) reinforced by the management option that drain needle leaved forest when they are planted on less draining soils. Overall, Rainfall has a strong highly significant positive impact on specific volume and remains the main input of the streamflow.

Despite the negative effect of forest cover on streamflow magnitude, this study shows a significant positive effect on the specific discharge exceeded 95% of the time, annually and during the vegetation season. This is a sign of a positive impact of forest on the water supply in low flows conditions which is of extreme importance regarding riparian and aquatic habitat provision. This can also be directly linked to the water quality as water dilutes nutriments and pollutants but also regarding stream temperature. However, this is out of the scope of this study. Low variability was described by the models used to study BFI (R^2 of 0.22 for the annual model and around 0.30 for the seasonal models). This testifies the importance of other effects than land cover in explaining baseflow such as highlighted by Price (2011). Rainfall has a highly significant strong negative impact on the BFI which is expected since the BFI is the proportion of baseflow to total streamflow over a continuous period of record. In these models no significant effect of forest (PC1) nor PC2 is shown whether annually or during the vegetation period. Literature review does not provide us with strong assumptions of what we would expect in an “ideal” experimentation comparing numerous catchments while controlling other factors than land cover. Indeed some studies show a positive effect of forest on this indicator in accordance with the better infiltration of forested soils (Bruijnzeel, 2004; Price et al., 2011), while others show the reverse effect linked to higher evapotranspiration rates (Hicks et al., 1991). Furthermore the differential impacts of forests compared to grasslands are less clear than with other land uses such as conventional crops regarding vegetation (Granier, 2007) and obviously artificial surfaces. The "year" effect also has a highly significant impact on the 3 studied indicators of water supply. We assume that this effect is a combination of several factors such as climate variables and particularly temperature conditions.

3.5.1.3 Water damage protection

We adopted the specific discharge exceeded 5% of time and the flashiness index as hydrological variables reversely linked to the flood protection service. Regarding the specific Q_{05}, interestingly and unlike for the Q_{95}, we do not observe any positive significant effect of forest on this indicator. There is even a slightly significant negative effect of forest during the non-vegetation period. The flashiness index (FI) which compares the 5th and the 95th percentiles of discharges through a ratio is negatively impacted by forest cover which is a sign of relative stability of hydrological regime of the forested catchments. This can be linked to a positive impact of forest on flood protection service.

3.5.2 Strengths / Limitations of the methods and Perspectives

The main strengths of the proposed method can be described as follows: (i) to characterise ecosystem services at the catchment scale we chose indicators easily extractable from broadly available data sets (discharge data series) with high frequency monitoring (in our case 10 min steps discharge data); (ii) we applied a simple and commonly used methodology entirely implemented in a statistical open source software (R) making this analysis easily replicable in other regions and/or through time; (iii) this method allowed us to show effect of land cover on hydrological services but also provided us with broader understanding of factors influencing ecosystem functions further influencing ES. Some limitations of the study can also be pointed out: (i) compared to classical experimental pair-wised approaches, working with “real-life” catchments actually monitored by the public administration complicates the learnings that can be drawn from the study. Indeed, other factors correlated with forest cover impact hydrological services (e.g. slope, soil infiltration capacity, tree species water use, phenology…). Nevertheless, this approach allowed us to provide information about ES at the landscape level. (ii) the selected indicators of hydrological flows are statistics characterizing the hydrogram overall, further research could concentrate on specific rainfall events and further detail the behaviour of the catchment to provide insight at the event scale of the effect of forest cover on the flood protection service for instance.

3.5.3 Land cover proxies for ES assessment?

Regarding the debate of land cover proxies use in assessing ES, this study highlights that for the hydrological ES considered here, other factors than land cover impact water flows at the catchment scale. For example, the analysis of the effect of the type of forests (PC2) on water supply (specific volume) suggests an effect of terrain topography but also soil types and – we assume – related forest management options (artificial drainage). Regarding BFI and knowing that the absence of non-linear relationship was checked, low R^2 of the model showed that there were obviously other important factors acting on this aspect of water supply. The relative proportion of forest within the catchment could also be part of the explanation: even if forests are the dominant land cover, their effects on hydrological indicators may be dampened by the effects of other land covers. Another factor is the type of forest, in particular the differences between needle-leaved and broad-leaved species, which induces different seasonal effects on catchment water balance. In spite of this, effects of forests on water supply could be shown: overall negative effect on specific volume but positive effect in low flow periods (Q_{95s}).

Regarding flood protection the study did not show any effect of forest on high flows (Q_{05s}) whereas forest cover showed a negative impact on the flashy behavior of the catchment. In this context we recommend further research integrating at best local condition factors (soil characteristics, slopes, etc.) where each land cover is actually located (and not in the catchment overall) in order to come up with integrative proxies indicators of ecosystem services. In the case of hydrological services the effect of ‘year’ (i.e. climatic characteristics) and rainfall were highly significant in most of the cases showing the importance of climatic condition on ES. In the current context of climate change, inducing more frequent spring and/or summer droughts, this draws attention to the adverse impacts it may generate towards water related ES.

Chapter 4 Forest cover impact on instream water supply in terms of physico-chemical water quality

The following text is directly taken from the following published article:

Brogna, D., Michez, A., Jacobs, S., Dufrêne, M., Vincke, C., Dendoncker, N., 2017a. Linking Forest Cover to Water Quality: A Multivariate Analysis of Large Monitoring Datasets. Water 9, 176.

As for Chapter 3, a preliminary section first remind the thematic and methodological objectives of the PhD pursued in this study. Then, precisions with regard to the published version are presented.

Preamble and precisions

This research main objective is the **study of the impact of forest cover on instream water supply in terms of physico-chemical water quality**. In this study, we approach the physico-chemical water quality through nine variables (i.e. dissolved oxygen, dissolved organic carbon, pH, total phosphorus, ammonium, nitrites, nitrates, chloride and sulphate concentrations). We quantify forest cover and independent effect of forest cover types (i.e. needle-leaved and broadleaved forests) on physico-chemical water quality in comparison to other LULC at the regional scale (10 years dataset). We assess the temporal variability of this effect by testing annual and seasonal effects.

Transversal methodological objectives are in here attained, through the development of a replicable approach (i) based on easily accessible data, monitored in many countries (i.e. EU Water Framework Directive monitoring datasets), (ii) using multivariate statistical methods and (iii) with main processes run in open source statistical software. The enlargement of scope of the derived results aiming to come up with land planning recommendations is reached through the study of 362 monitoring stations spread across the region whose upstream catchments have mixed land covers.

Precisions:

In the following text, the word 'nutriments' should be replaced by nutrients, and the section number mentioned in the last sentence of section 4.3.2 should be 4.3.3 instead of 3.3.

Caption in Figure 4-2 should be complemented by : (source : TOP10VGIS, NGI). However, as the paper has been published as is, we prefer to leave the text unaltered.

Abstract

Forested catchments are generally assumed to provide higher quality water. However, this hypothesis must be validated in various contexts as interactions between multiple land use and land cover (LULC) types, ecological variables and water quality variables render this relationship highly complex. This paper applies a straightforward multivariate approach on a typical large monitoring dataset of a highly managed and densely populated area (Wallonia, Belgium; 10 years' dataset), quantifying forest cover effects on nine physico-chemical water quality variables. Results show that forest cover explains about one third of the variability of water quality and is positively correlated with higher quality water. When controlling for spatial autocorrelation, forest cover still explains 9% of water quality. Unlike needle-leaved forest cover, broad-leaved forest cover presents an independent effect from ecological variables and explains independently 4.8% of water quality variability while it shares 5.8% with cropland cover. This study demonstrates clear independent effects of forest cover on water quality, and presents a method to tease out independent LULC effects from typical large multivariate monitoring datasets. Further research on explanatory variables, spatial distribution effects and water quality datasets could lead to effective strategies to mitigate pollution and reach legal targets.

4.1 Introduction

Water is the most essential component for the life of all beings (Haddadin, 2001; UN-Water, 2014). However, freshwater systems and consequently human well-being are directly threatened by human activities (Meybeck, 2003; Millennium Ecosystem Assessment, 2005a; Vörösmarty et al., 2010). In response to the global degradation of ecosystems and their services, water quality management is at the core of policies such as the European Water Framework Directive (Directive, 2000/60/CE) aiming at maintaining or restoring the chemical, physical and biological integrity of surface and groundwater bodies. Managing water quality is challenging and implies to deal with both point and non-point source pollutions. As non-point source pollutions result from complex runoff and landscape interactions, they are more complex to identify than confined point source pollutions (Carpenter et al., 1998). Land use and Land cover (LULC) are key landscape elements affecting water quality through their impact on non-point source pollution.

Previous studies attempting to address LULC impact on water quality broadly correlate urban and/or agricultural LULC with poor water quality either at the catchment or riparian scale. These represent water quality through several variables, but nitrate and phosphate, which are at the basis of eutrophication problems, are the most studied. More specifically, Álvarez-Cabria et al. (2016) model three water quality variables (temperature, concentrations of nitrates and phosphates) in a watershed located in Spain. Their results show that nitrate and phosphate concentrations were mainly related to agricultural LULC and urban LULC, respectively. Chen et al. (2016) show that urban land is the dominant factor influencing nitrogen, phosphorus and chemical oxygen demand in highly urbanized regions of a catchment located in eastern China but that agricultural land has the greatest influence on nitrogen and phosphorus in suburban and rural areas. Yu et al. (2013) also show direct and indirect negative impact of urbanization and agricultural activities on water quality in an urban area of China. De Oliveira et al. (2016) assess LULC effect on nitrate, total ammonia nitrogen, total phosphorus and dissolved oxygen in a Brazilian watershed. Their results (correlations) point out that urban areas and agriculture/pasture tend to worsen water quality while some models (i.e., nitrate and total phosphorus) were not valid. Hwang et al. (2016) show that relationships between urban LU and water quality vary according to the degree of urbanization. These studies are often specific to one or few catchments, treat different water quality variables at different temporal and spatial scales but broadly show negative impact of urban and agricultural LULC on water quality. These results are often presented in opposition to forest cover associated with higher water quality.

Forest is one of the LULC that interacts the most with water, whether in terms of quantity or quality, and consequently has an impact on hydrological services—which can be grouped into water supply and water damage mitigation and viewed in terms of quantity, quality and timing (Brauman et al., 2007; Brogna et al., 2017b; Carvalho-Santos et al., 2014). Indeed, forests and forest soils alter, relative to other land uses and soil types, each of the five physical, chemical and biological functions involving the reception, processing and transfer of water(Neary et al., 2009). Forest general impact on water quality, relative to other land uses, can be summarized as water with less sediments and water with fewer nutriments (Neary et al., 2009; TEEB, 2010). Several studies state that forested catchments tend to have more stable water quality conditions (Fiquepron et al., 2013; Łowicki, 2012; Tong and Chen, 2002) but we did not come across any study that quantifies this impact on several pollutants simultaneously, on a large scale and over a relatively long period. In addition, one may question the validity of these hypothesized relationships under different latitudes, at different temporal and spatial scales, under various management types, according to different forest types (i.e., needle-leaved vs. broad-leaved forests). In addition, global changes affect water quality and quantity and question the assumption that studies from last century can be used to face future conditions (Vose et al., 2011). Finally, Giri and Qiu (2016) stress the importance of assessing the relationship between LULC and water quality (see also Chauhan and Verma (2015)), pointing out that improving our understanding of these can help managing water quality in unmonitored watersheds but also that this knowledge can provide guidelines to watershed managers and policymakers in the land planning decision process.

In Wallonia (Belgium), few studies attempting to assess the impact of LULC and in particular forest cover on water quality were published. Some specific studies related to the subject exist as the assessment of variability of nitrate removal in riparian zones (Dhondt et al., 2006) or the in lab assessment of temperature, throughfall volume and ammonium cation deposition impact on soil solution nitrate concentrations, nitrous oxide emissions and numbers of ammonium oxidisers from a forest stand soil (Carnol and Ineson, 1999).

Regarding nitrates, Sohier and Degré (2010) present a hydrological model for evaluating the effectiveness of agricultural policy measures on nitrate concentration in surface and ground waters.

Different methods can be used to assess the relationship between water quality and LULC. Giri and Qiu (2016) classified these into three categories: monitoring, hydrologic/water quality modeling and statistical modeling. Direct and real-time monitoring in the stream is expensive, time consuming, and ineffective for larger areas. Hydrological and water quality models used in several studies (Carvalho-Santos et al., 2015; Jomaa et al., 2016; Lin et al., 2015) require vast amounts of data, are complex, costly and time consuming to calibrate, and therefore only applied on one single or few catchments. Statistical methods tend to be simpler, easier to apply, and more efficient than physically-based hydrologic/water quality models when observed data are limited in time and when datasets are covering many different catchments (Wan et al., 2014).

Regarding this context, this study applies multivariate statistical methods to mine a regional monitoring dataset from the highly managed and densely populated Walloon region (Belgium). It provides a quantification of forest cover effect on several physico-chemical variables simultaneously. More specifically, this paper:

i. Analyzes the link between sub-catchments' LULC and the legal status of in stream water quality

ii. Quantitatively assesses the link between forest cover and nine water quality variables, verifying spatio-temporal variability;

iii. Quantifies the independent effect of forest cover types (i.e., needle-leaved and broadleaved forests) on water quality relatively to effects of other LULC.

This study develops a novel approach, replicable in space and time, based on easily accessible public data (public monitoring network data directly linked to the Water Framework Directive and LULC data), and using powerful but simple statistical methods ran in open source statistical software (R stat, (R Core Team, 2013)).

4.2 Materials and Methods

4.2.1 Study area

The study area is the southern region of Belgium (Wallonia) covering 16,902 km2 (ca. 55% of Belgium's area, see Figure 4-1). We work on the publically managed river network and in particular on 362 water quality stations monitored for the Walloon Public Service (WPS, (SPW - DGO3, n.d.), Figure 4-1).

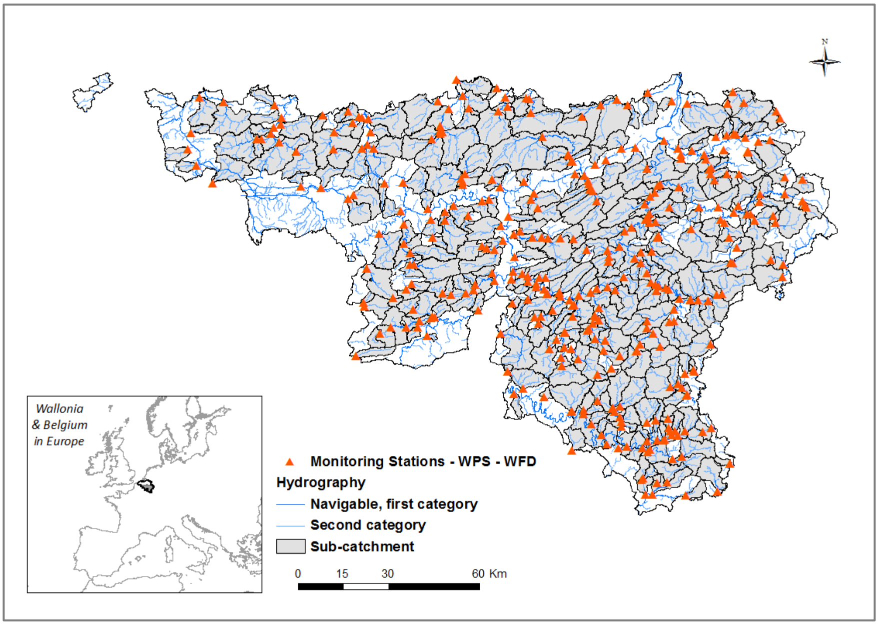


Figure 4-1. Water quality monitoring stations used in this study and sub-catchments

Population density in Wallonia is 202 inhabitants per square kilometer and, with hardly less than 1% of the territory beneficiating of a natural reserve status, all landscapes are mostly managed or perturbed by human activities. Main LULC are agriculture land (52%) comprising grassland (30%) and cropland (22%); forests (30%) split into needle-leaved (13%), broad-leaved (16%) and mixed forests (1%); and artificial surfaces (10%) (see LULC dataset, Section 2.4 and Figure 4-2). Cropland cover is mostly located in the North of the region and at a low elevation while forests (especially needle-leaved forests) are located in the South at a higher elevation. In the studied sub-catchments (see delineation method below), the most represented LULC classes are forests, grassland and cropland (see boxplot in Supplementary Materials Figure S1).

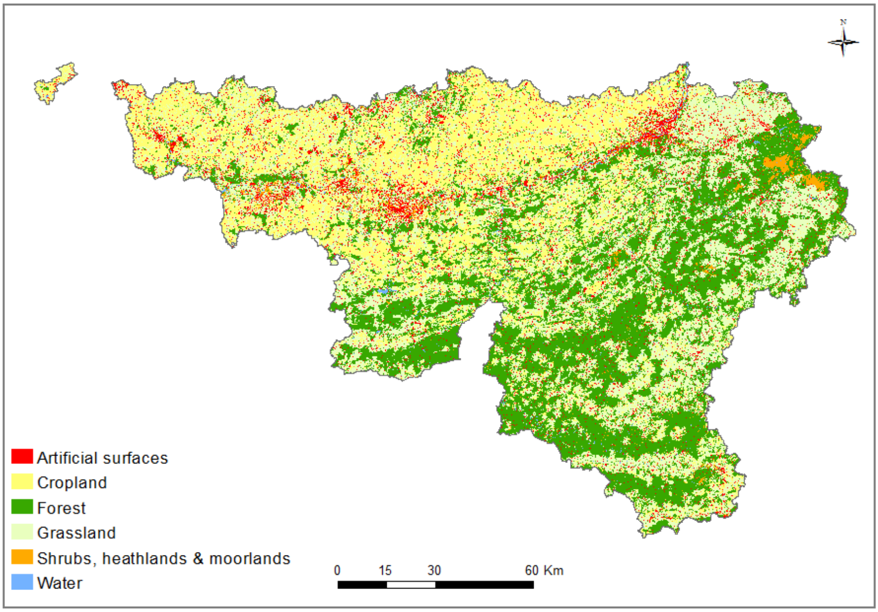


Figure 4-2. Land use and land cover (LULC) in Wallonia

Intensive agriculture impacts water quality through the use of mineral fertilizer, in particular nitrogen (N) and phosphorus (P), causing eutrophication and drinking water quality degradation. Even if declining since 1990, inputs of nitrogen and phosphorus were still above the European average in 2001 (SPW-DGO3-Direction de l'Etat Environnemental, 2014, 2007). Nitrogen still exceeded (about double) the European average in 2012 while phosphorus decreased to around half of the European average.

Regarding species composition, broad-leaved forests are largely dominated by oaks (*Quercus robur* and *Q. petraea*) and beech (*Fagus sylvatica*) but other species such as birch (*Betula pendula*), ash (*Fraxinus excelsior*), maple (*Acer pseudoplatanus*), and hornbeam (*Carpinus betulus*) are also present (Alderweireld et al., 2015). Needle-leaved forests are very intensively managed with the use of exotic species (mainly spruce (*Picea abies*) but also

Douglas fir (*Pseudotsuga menziesii*), larches (*Larix sp.*), and pines (*Pinus sylvestris* and *P. nigra*)), in even-aged stands conducted with systematically clear-cuttings, and high drainage infrastructure on wet soils. In the last century, forests in Wallonia have suffered from large inputs of sulfur and nitrogen through acidic rainfalls. Even if this phenomenon has been declining since 1990, it affected forests during the studied decade (SPW-DGO3-Direction de l'Etat Environnemental, 2014).

4.2.2 Workflow

The overall methodology is illustrated in Figure 4-3 and described below. The first section describes the physico-chemical water quality dataset processing. Secondly, the delineation of sub-catchments and LULC characterization are explained. Finally, the methodology to assess the link between forest cover and water quality is presented. In this last part, we detail the following analyses: the link between forest cover and WFD standards, the quantification of the forest cover effects on water quality and spatio-temporal aspects, and, finally, the partitioning of the LULC effect on water quality between forest types and LULC and their shared effects while controlling for ecological gradient.

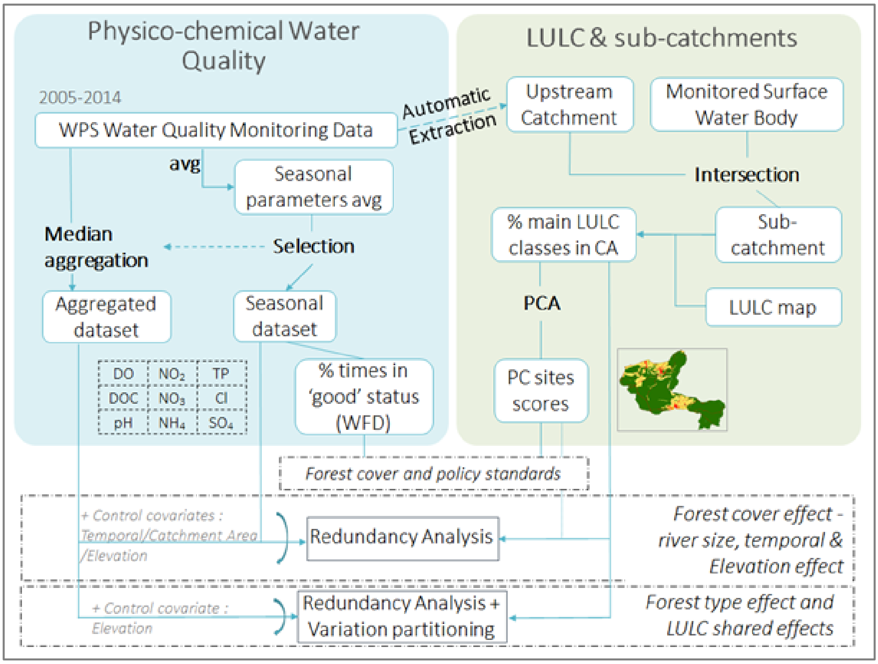


Figure 4-3. Schematic figure of the methodological approach to link forest cover and water quality. With WPS: Walloon Public Service, avg: average, WFD: Water Framework Directive, PCA: Principal Component Analysis

4.2.3 Physico-chemical Water Quality

Physico-chemical water quality was studied through selected variables (Table 4-1) from the monitoring performed by the WPS. We processed data from 10 hydrological years (October 2004 to September 2014). The hydrological year—i.e., the annual base period—is temporally defined by the period from October to the following September and corresponds to the natural hydrological cycle (Gailliez, 2013).

We selected 362 stations from the monitoring network dataset of the WPS. The selected stations are characterized by an upstream catchment that can be extracted automatically from a digital elevation model. Consequently, artificial water bodies such as artificial canals hydrologically disconnected or crossing watersheds were excluded. Moreover, we excluded stations whose upstream catchments are partially located outside Wallonia. In order to maximize the number of multiple observations and still fit to seasonal vegetation development, we averaged water quality variables' values by season and by station. Following Brogna et al. (2017b), seasons were delineated according to vegetation development and rainfall distribution, splitting the year into a "non-vegetation season" (October–March) and a "vegetation season" (April–September). Among the water quality variables monitored by the administration, variables were excluded: (1) if the Pearson correlation coefficient exceeded 80% to exclude highly redundant variables for multivariate analysis purposes (Olsen et al., 2012); or (2) if missing data exceeded 5% of the dataset (not to lose too many sampling dates in the multivariate table). Thus, 9 out of 16 variables were analyzed in this study (Table 4-1).

Table 4-1. Physico-chemical variables investigated.

Quality 'group'	Variable		Unit
Oxygen balance	Dissolved Oxygen	DO	(mgO_2/l)
	Dissolved Organic Carbon	DOC	(mgC/l)
Phosphorus	Total phosphorus	TP	(mgP/l)
Nitrogenous material	Ammonium	NH_4	(mgN/l)
	Nitrites	NO_2	
	Nitrates	NO_3	
Acidification	pH	pH	-
Mineralization	Chloride	Cl	(mg/l)
	Sulphate	SO_4	(mg/l)

The seasonal dataset consists of a 9 variables × 3793 observations (related to 362 stations) table. We excluded values that exceeded the 99th quantile as they are outliers representing incorrect values. We applied Log- or square-based transformations when improving the normality of variables' distribution was needed. We computed the percentage of times the monitoring station was classified as "good status" (i.e., good or high status according to WFD) and linked this legal status to land cover. This dataset was also used as input to study the link between forest cover and water quality and test the influence of the stream size effect and temporal (seasonal and year) effect.

Following the results of the temporal effect analysis, we built an aggregated dataset. We aggregated variables' values by station over the entire 10-year dataset using two aggregation function types: the 90th quantile (and 10th quantile for the dissolved oxygen variable) and the median values. As results were highly similar, we only present the median aggregation results. Again, we applied Log- or square-based transformations improving the normality of variables' distribution was needed. We used this dataset to quantify the independent effect of forest cover and forest cover types (i.e., needle-leaved and broadleaved forests) on water quality relatively to effects of other LULC.

4.2.4 **Land use and land cover data**

Different spatial units can be considered in order to study the impact of land cover on water quality study. Some authors consider the entire upstream catchment as the spatial unit of LULC reference (Tu, 2013). Others use the riparian zone (i.e., a buffer around the stream) to characterize the land cover impacting water stream quality (Li et al., 2009). Both approaches have some drawbacks. Indeed, when considering the upstream catchment in non-spatial statistical methods, the same importance is given to points irrespective of their distance to the monitoring station ignoring processes such as the self-purification of the stream. On the other hand, Giri and Qiu (2016) point out problems with the riparian zone approach such as the absence of a uniform way to define its width or the fact that this zone does not represent nor behave ideally in terms of hydrological variation in the landscape. Pratt and Chang (2012) also state that while riparian land cover affect water quality, a wider contributing area must be included to take into account distant sources of pollutants.

In this study, the area associated with the monitoring station is the intersection of the upstream catchment station (automatically extracted from the ERRUISSOL digital terrain model http://geoportail.wallonie.be) with the monitored surface water body defined by the WPS. This spatial unit of reference is thus directly based on the Walloon surface water body delineation which is the basic unit area for water quality assessment in line with the WFD directive (SPW-DGO3-Direction de l'Etat Environnemental, 2016). This water body delineation already integrates different variables as catchment size and hydromorphological parameters. In addition, this choice overcomes some above-mentioned drawbacks while providing a clear "rule" to apply across the whole region. This spatial unit of analysis will be further referred to as “sub-catchment”.

We used the Top10VGIS land cover data set provided by the Belgian National Geographic Institute (NGI, www.ngi.be) to characterize the land cover of the sub-catchments. This vector data set covers the whole of Belgium. It is based on the NGI topogeographic data that classify LULC into 37 classes. For the purpose of our study, we either selected the original land cover classes as such or aggregated them to end up with seven classes of interest: needle-leaved forest, broad-leaved forest, cropland, grassland, artificial surfaces, water surfaces, shrubs–heathlands and moorlands. We computed percentages of these classes in each sub-catchment. Following Brogna et al. (2017b), we assumed that the evolution of the retained classes in the region was minor throughout the studied decade.

To relate the LULC to water quality, we intersected the Top10VGIS dataset with each sub-catchment. To control the natural correlation between percentages of LULC variables, we constructed independent LULC variables by running Principal Component Analysis (PCA) on the main land cover classes' percentage in the sub-catchments (i.e., needle-leaved forest, broad-leaved forest, grassland and cropland). We ran these PCA on standardized data, and extracted the stations coordinates on first and second components as independent LULC variables (i.e., LULC1 and LULC2).

4.2.5 Coupling forest cover and water quality

First, we performed a preliminary analysis on the seasonal dataset to assess the link between the main LULC of the region and the percentage of time over the decade each monitoring station was classified as “good status” according to current standards (SPW-DGO3-Direction de l'Etat Environnemental, 2016).This provided a comprehensive picture of the relationships between LULC and policy standards for each water quality variables group (see Table 4-1). A station was considered in “good status” if it was classified as such for every component of the water quality variables group (see Table 4-1).

Then, we quantified the relationship between forest cover and physico-chemical water quality by applying a redundancy analysis (RDA, see Legendre and Legendre (2012c), R package vegan (Oksanen et al., 2017)) on LULC independent variables resulting from PCA (see above). This multivariate analysis allows capturing the linear relationship between dependent variables (i.e., physico-chemical variables, referred to as WQ in equations) and a matrix of explanatory variables (i.e., sub-catchments land cover variables). This analysis thus quantifies the percentage of water quality variability explained by LULC variables (Oksanen et al., 2017). It allows quantifying and excluding the variability explained by certain covariates (e.g., season, year, upstream catchment area). We ran these RDA on centered and scaled variables because of the heterogeneity of the water quality variables units.

Analysis of the seasonal dataset explains the WQ matrix by the linear combination of the main land cover classes of interest:

$$WQ \sim LULC1 + LULC2 \quad \text{[Equ 4. 1]}$$

where WQ = matrix of physico-chemical measurements (see Table 4-1); and LULC1 and LULC2 = independent LULC variables derived from the land cover PCA (i.e., sites scores on the first two axes of the LULC PCA).

However, following the interpretation of this first RDA, we simplified both the analysis and the results' reading by applying an RDA directly on the percentage of forest cover in the sub-catchments (see Section 3.2).

$$WQ \sim Forest \quad \text{[Equ 4. 2]}$$

where WQ = matrix of physico-chemical measurements (see Table 4-1), and Forest = percentage of forest cover in the sub-catchment (i.e., the sum of needle-leaved, broad-leaved and mixed forest percentage).

We tested the effect of the river size, directly linked to the discharge, to verify our choice to work with pollutant concentrations rather than loads, by putting it as covariate in the partial RDA under the proxy of whole upstream catchment area.

$$WQ \sim Forest + Condition\ (upstream\ catchment\ area) \quad \text{[Equ 4. 3]}$$

We also tested the impact of temporality while considering the season, year and their interaction as covariates.

$$WQ \sim Forest + Condition\ (season + year + season \times year) \quad \text{[Equ 4. 4]}$$

Following the analysis of the temporal variability impact, we ran the same analysis (Equation (4.2)) on aggregated water quality values (median value by station over the 10 years).

As spatial autocorrelation often explains an important part of biological structures (Legendre, 1993), we partially controlled it by using "elevation" as covariate when quantifying the forest cover effect on water quality (Equation (4.5)). Indeed, there is a strong continuous ecological gradient in Belgium, that is highly correlated to elevation (Dufrene and Legendre, 1991; A Noirfalise, 1988). Dufrêne and Legendre (1991) show that elevation, although not exceeding 700 m in Belgium, explains almost all the geographic structure of several ecological variables given their spatial autocorrelation.

$$WQ \sim \text{Forest} + \text{Condition (Elevation)} \qquad \text{[Equ 4. 5]}$$

Finally, we distinguished effects from needle-leaved and broad-leaved forest covers while capturing the shared effects between LULC classes and distinguishing those from the elevation effect. We therefore ran several RDA with elevation as covariate and a variation partitioning between most important LULC and elevation (Legendre and Legendre, 2012c). The latter method divides the explained variance of a dataset between partial effects (i.e., the proportion of variation explained by a particular variable) and shared effects (i.e., variation that cannot be attributed to one variable but is shared between two or more).

4.3 Results

4.3.1 Forest cover versus legal standards

Figure 4-4 shows the biplot of the PCA ran on the main land cover classes' proportions at the sub-catchment level. The first two components explain 80% of the variability of the data set. The first component (LULC1), which explains 47.6% of the dataset variability, opposes grassland and cropland on the one hand (negative side), and needle-leaved and broad-leaved forests on the other hand (positive side). The second component (LULC2) explains 32.4% of the dataset variability and is mostly based on an opposition between grassland (positive side), and broad-leaved forest and cropland classes (negative side). Figure 4-4 illustrates the percentage of times that each monitoring station was classified in "good status" throughout the studied decade following current WFD standards (seasonal values). Except for the "mineralization" group (i.e., sulfates and chlorides) for which all stations are 100% of the time in "good status", we observe a clear gradient linked to LULC. Regarding nitrogenous material (i.e., nitrates, nitrites and ammonium), the majority of stations that, most of the time, do not reach the "good status" have high cropland or grassland cover. Moreover, very few stations with high cropland cover are classified as "good status", even for part of the decade. The same trend—even if less strong—is observed for the total phosphorus standards. Regarding oxygen balance (i.e., dissolved oxygen and dissolved organic carbon), stations that, most of the time, do not reach the "good status" are linked to high cropland cover (with a relatively important grassland cover).

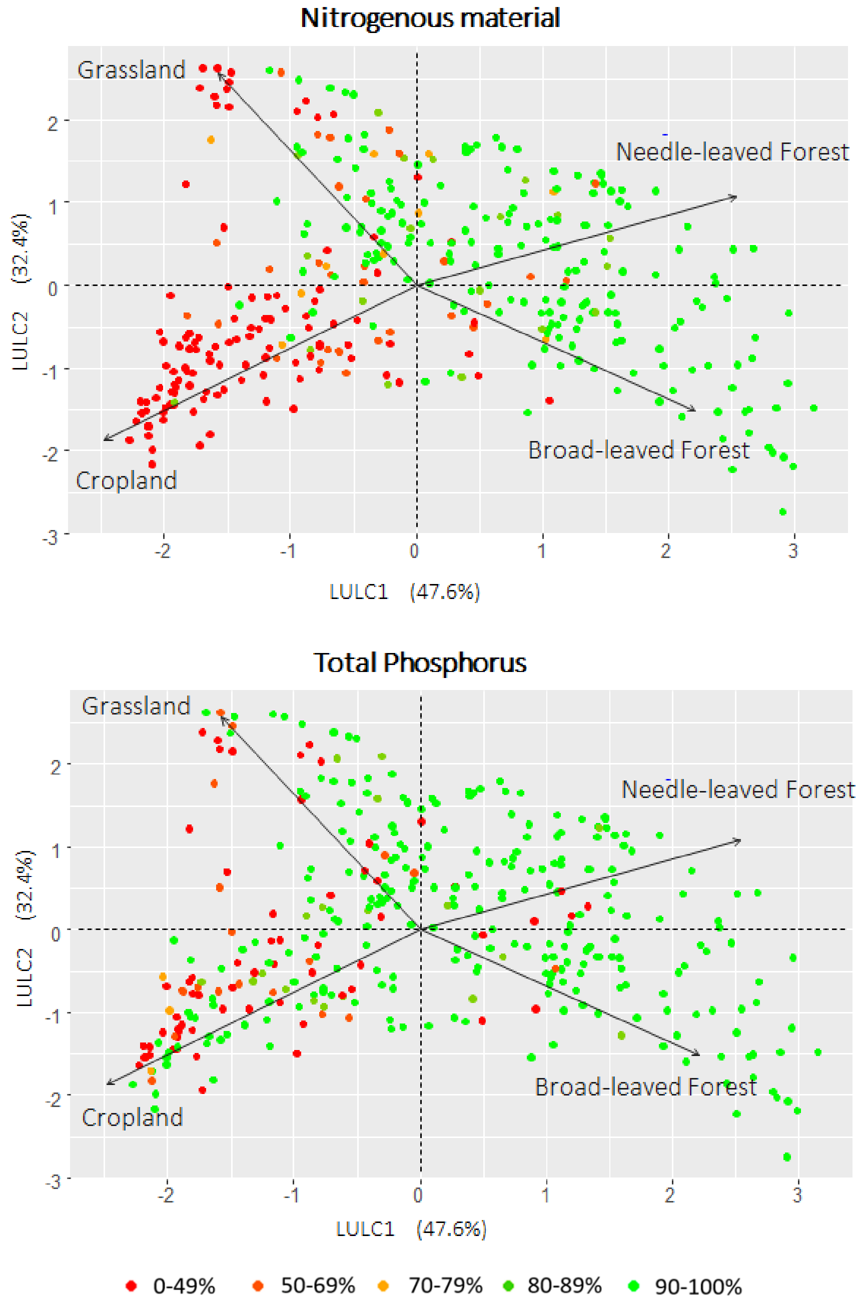


Figure 4-4. Biplot representing the monitoring stations from a PCA on the main land cover classes in studied sub-catchments. Colors represent the percentage of time the station was classified in 'good status' according to current Water Framework Directive standard

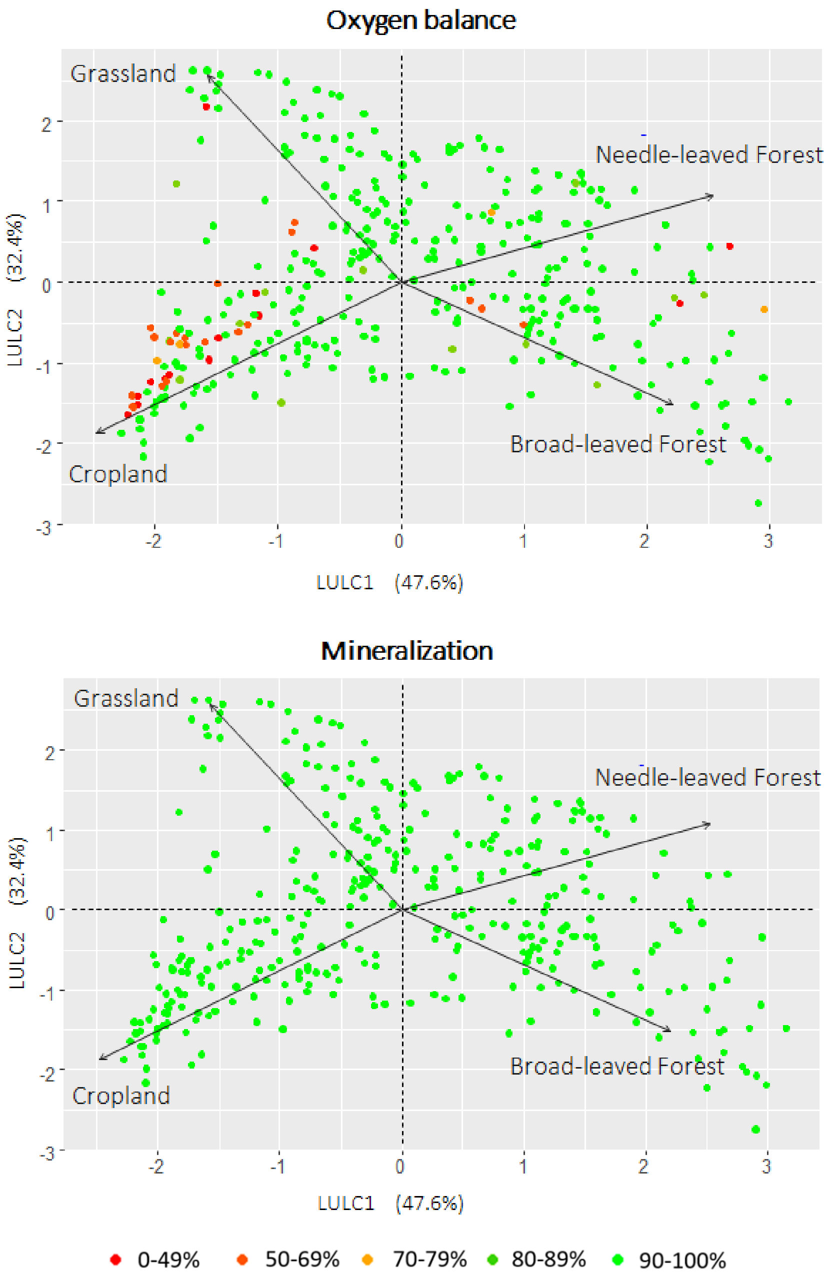


Figure 4-4 (end): Biplot representing the monitoring stations from a PCA on the main land cover classes in studied sub-catchments. Colors represent the percentage of time the station was classified in 'good status' according to current Water Framework Directive standard

4.3.2 Forest cover versus multivariate water quality

Redundancy analysis on seasonal water quality values (Equation (4.1)) and LULC independent variables (see Figure 4-4) showed that the first constrained axis explains 43% of the water quality variability, and the second one explaining less than 1% of the water quality variability. This first axis is highly correlated to forest cover in sub-catchments. Consequently, following RDA in this study are directly based on forest cover percentage in sub-catchments. Table 4-2 (seasonal dataset) shows the percentage of variability explained by forest cover in these RDA either considering the forest cover only or removing some effects through the use of covariates. Temporal, river size and elevation effects have thus been removed successively. We applied permutation tests to each model produced in this study and all the presented models are significant.

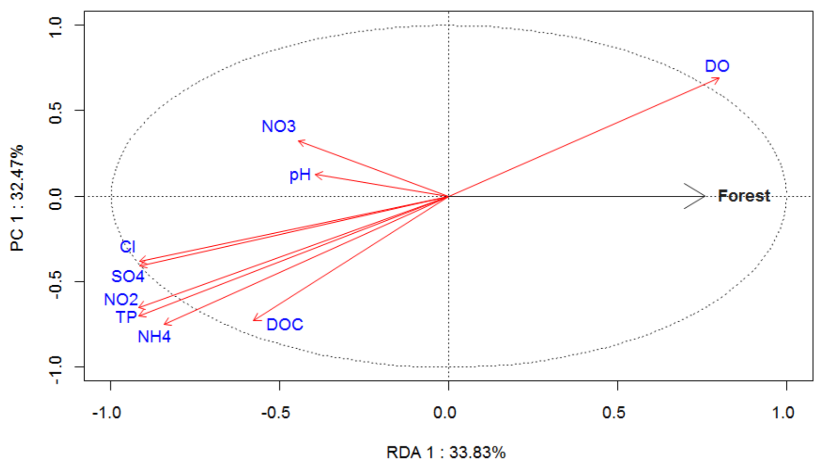
Table 4-2. Redundancy analysis results for seasonal dataset (Oct-Mar, Apr-Sept) and aggregated dataset (median value over the study period): variability (%) explained by forest cover, by covariates and shared effects

Redundancy analysis covariate(s)	Variability explained by forest cover (%)	Variability shared between forest cover and covariate(s) (%)	Variability explained by covariate(s) (%)
Seasonal dataset			
No covariates - [Equ 4.2]	29.6	-	-
Covariate : upstream catchment area - [Equ 4.3]	29.3	0.3	0.5
Covariates : season + year + season*year - [Equ 4.4]	28.4	1.2	7
Aggregated dataset			
No covariates - [Equ 4.2]	33.8	-	-
Covariate : Elevation - [Equ 4.5]	9.3	24.5	12.9

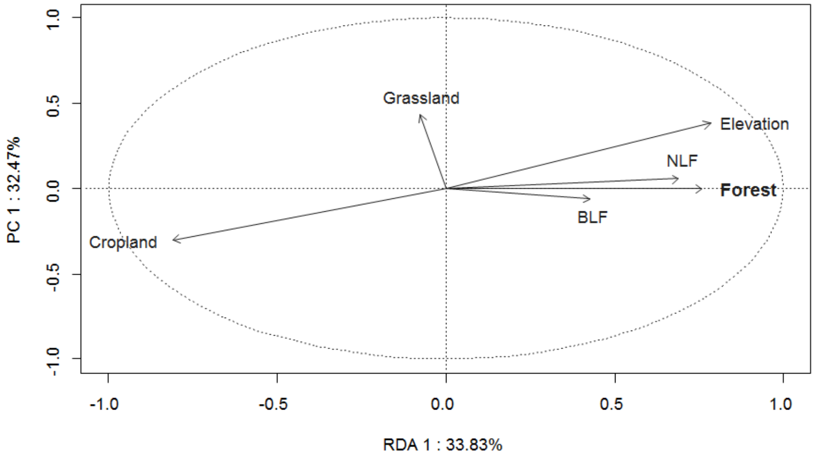
Forest cover explains 29.6% of the seasonal physico-chemical water quality. The upstream catchment area effect is independent from the forest cover effect (shared effect of 0.3%) and explains 0.5% of the total variability of water quality. Temporal effect is also independent from the forest cover effect (shared effect of 1.2%) and explains 7% of total water quality variability. Consequently, we present below, and in the second part of Table 4-2, results from the aggregated dataset.

Results of RDA (Equation (4.2)) on the aggregated dataset show that forest cover explains 33.8% of the water quality variability (see Table 4-2, aggregated dataset). Figure 4-5 A–C presents these results in a factorial plan constituted by the constrained axis (RDA1) and the first residual axis (PC1).

A



B



C

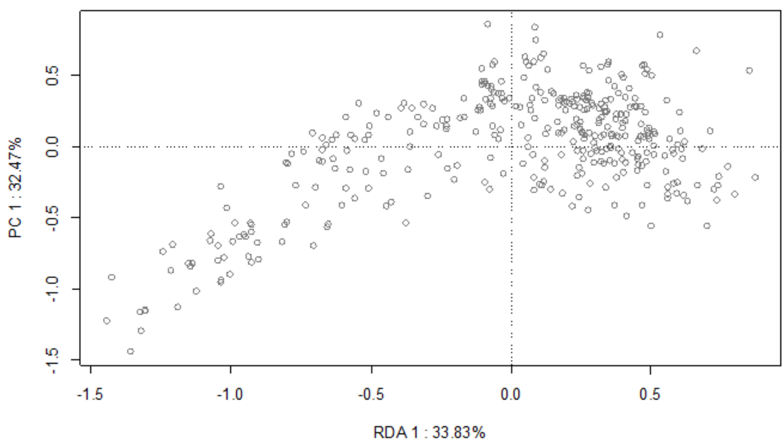


Figure 4-5. RDA results showing the link between forest cover and water quality. X-axis represents the constrained axis and y-axis, the first residual component. (A) Correlation circle of the RDA with 'active' variables and the constrained variable (forest cover), with DO: Dissolved Oxygen, NO3: Nitrates, Cl: Chloride, SO4: Sulfates, TP: Total Phosphorus, NO2: Nitrites, NH4: Ammonium and DOC: Dissolved Organic Carbon. (B) Correlation circle of the RDA with constrained variable (forest cover) and 'passive' variables (with NLF: Needle-leaved forest and BLF: Broad-leaved forest). (C) Stations location in the same plan.

Figure 4-5 A shows the correlation circle of the RDA with "active" variables (water quality variables) and the constrained variable (forest cover percentage in sub-catchments). Forest cover is clearly linked to high stream water quality. Indeed, Dissolved Oxygen is positively correlated to forest cover while Ammonium, Nitrites, Total Phosphorus, Sulfates and Chloride concentrations are highly negatively correlated with forest cover. Dissolved Organic Carbon, pH, and Nitrates are also negatively correlated to forest cover but to a lesser extent. Figure 4-5 presents the correlation circle from the RDA with both the constrained variable (i.e., forest cover) and "passive" variables (i.e., LULC classes and elevation). The constrained variable is obviously positively correlated with needle-leaved forest cover and broad-leaved forest cover in a less extent and inversely correlated to cropland cover. Grassland cover hardly contributes to the constrained axis. Figure 4-5 C shows the position of stations in the same factorial plan. Elevation is also correlated with this constrained axis and RDA (see Table 4-2, aggregated dataset) shows that forest cover still explains 9.3% of water quality variability when the shared effect with elevation is removed. This is taken into account and refined in Section 3.3.

4.3.3 Independent and shared effects of LULC classes (included forest type distinction)

Results of RDA explaining water quality dataset with, on one the hand, each LULC class as constrained variable and, on the other hand, the same analysis but controlling for spatial autocorrelation (elevation as covariate) are presented in Table 4-3. Regarding forest types effect, needle-leaved forest cover explains water quality much more (29.3% versus 12.1%) than broad-leaved forest but is highly correlated with elevation, as is cropland cover. Indeed, when removing shared effect of needle-leaved forest cover with elevation, the explained proportion drops from 29.3% to 2.3%. On the contrary, broad-leaved forest cover effect is stable and independent from the ecological gradient; this LULC class explaining the most (10.9%) when removing the elevation effect. As expected (see Figure 4-5B), grassland cover hardly explains water quality variability.

Table 4-3. Independent effect of main LULC (with distinction between forest types) on water quality and effect when removing shared effect with elevation, F and p values of the models with (***): highly significant

	Independent effect				**With Elevation as covariate**			
Constrained variable- LULC (%) in sub-catchments	%	F	P		%	F	P	
Needle-leaved Forest	29.3%	148.5	0.001	***	2.3%	13.7	0.001	***
Broad-leaved Forest	12.1%	49.4	0.001	***	10.9%	75.2	0.001	***
Cropland	38.7%	226.1	0.001	***	7.9%	51.8	0.001	***
Grassland	1.2%	4.5	0.013	***	1.9%	11	0.001	***

Variation partitioning (see Figure 4-6) splits the variation of water quality dataset into independent effects of LULC classes and elevation (i.e., the proportion of variation that is not shared with other variables) and shared interactions (i.e., interaction that cannot be attributed to a single class). Needle-leaved forest and broad-leaved forest independently explain 4.8 and 1.8% of the total water quality variability, respectively. Broad-leaved forest cover shares 5.8% of explained variability with cropland but only 0.8% with elevation. An important part of variability (21.3%) is shared by needle-leaved forest, cropland cover and elevation, whereas less than 1% is shared by broad-leaved forest, cropland cover and elevation.

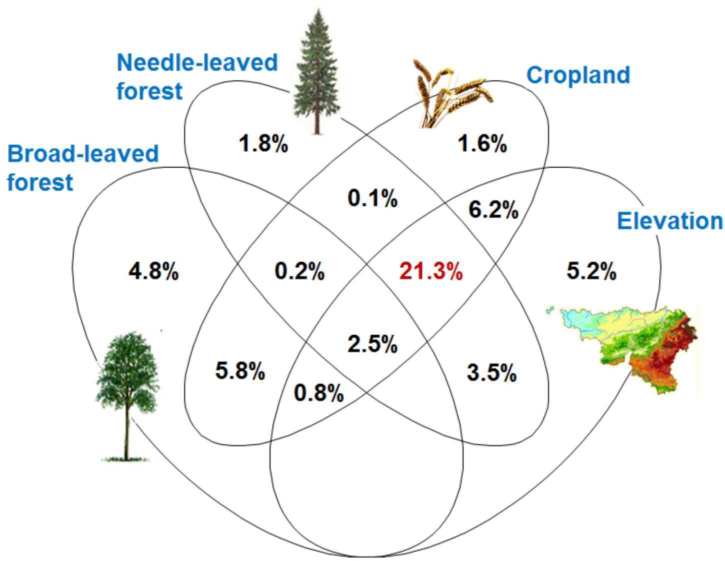


Figure 4-6. Venn diagram of the variance partitioning into main LULC classes and elevation. Figures are positive adjusted coefficients of determination and represent the variability explained by each subspace being either a single variable or shared effect between two or more variables.

4.4 Discussion

4.4.1 Forest cover and water quality

This study assesses the link between forest cover and water quality data at the regional scale, by applying straightforward multivariate statistics on a large monitoring dataset. The analysis of sub-catchment's LULC and the legal water quality status of streams (objective (i)) shows that sub-catchments with high forest cover tend to achieve "good status" over the studied decade more often than sub-catchments with high cropland and/or grassland covers. This is especially true for nitrogenous material and testifies that, despite the decrease in N input in agriculture since 1990, WFD target of "good status" is not yet fully reached. Sub-catchments with high grassland and cropland covers are also from far more polluted regarding phosphorus than forested sub-catchments. Nevertheless, stations with good phosphorus status are more frequent than in the case of nitrogenous materials. This analysis presents the advantage of providing an easily readable picture of the state of each station regarding water quality legal framework in relation with the LULC in its sub-catchment. This could provide a basis for further analyses to mitigate effects on water quality through improvement of catchment management. Indeed, some stations record unexpectedly high or low water quality in contrast to stations with similar LULC upstream signature. If an unexpected high water quality can be linked to particular practices or land planning measures, this could help managers to come up with locally relevant solutions.

Several insights were derived from the quantitative assessment of the link between forest cover and quality variables (objective (ii)). First, river size did not significantly affect the statistical relationship between LULC and water quality data. This allows aggregation of data from the entire monitoring network, with a high diversity of catchment sizes and thus discharges. It also justifies the use of concentrations instead of loads, unlike some authors such as de Oliveira et al. (2016) suggest. This renders the methodology less complex and more easily replicable on a such large numbers of sampling stations. Second, seasonal and between-year effects were insignificant across the full decade. This entails that the link between LULC and water quality reflects a background "multi-pollutants" load that can be considered as temporally stable. A potential seasonal effect as observed in other studies on particular relationships between particular variables and LULC (Álvarez-Cabria et al., 2016; Chen et al., 2016) might have been mitigated as the developed method treats several water quality variables together and as seasonal values are averaged values. For studies aiming to clearly focus on seasonal effects, we recommend specific and regular spatio-temporal sampling while focusing on homogeneous groups of pollutants regarding their seasonal variability (see e.g., Johnson et al. (1997)). Analysis of the aggregated dataset over the studied decade showed that forest cover explains one third of the median water quality variability. Using elevation effect as a proxy for various environmental variables and as a mean for controlling spatial autocorrelation, we demonstrated an independent forest effect of 9.3%. Specifically, in this densely populated region with highly managed landscapes and forests, sub-catchments which are dominated by forest, and have lower agriculture and grassland cover, provide water with higher oxygen availability and lower concentrations of Ammonium, Nitrites, Nitrates, Total Phosphorus, Sulfates and Chloride and Dissolved Organic Carbon. This is confirming previous main findings and reinforcing papers stating that forest cover is associated with higher water quality (Fiquepron et al., 2013; Łowicki, 2012; Tong and Chen, 2002). Main processes underlying these results are linked to the protection against erosion resulting in water with less sediments and fewer nutriments (Neary et al., 2009; TEEB, 2010) which is also due to lower pressures compared to agricultural and urban land uses.

Finally, using the power of both a large monitoring dataset and multivariate statistics, we quantified the partial effect of forest cover types (i.e., needle-leaved and broadleaved forests) on water quality and shared effects with other LULC and environmental variables represented by the elevation variable (objective (iii)). This study empirically confirms a clear effect, independent from elevation, of broad-leaved cover on water quality (10.9%). The important effects of needle-leaved forest (29.3%) and cropland cover (38.7%) are largely shared with elevation and can therefore not be proven as independent effects. Regarding shared effect between different LULC classes and elevation, our analysis shows that broad-leaved forest and needle-leaved forest independently explain 4.8 and 1.8% of the total water quality variability respectively. Broad-leaved forest cover shares 5.8% of explained variability with cropland but only 0.8% with elevation. The biggest part of the explained variability is shared by needle-leaved forest cover, cropland cover and elevation (21.3%). A part of this variability is surely linked to forest cover effect but cannot be attributed to it.

4.4.2 Strengths and limitations of the study

Some limitations of this approach can be pointed out: (i) the methodological design and data used do not allow for isolating quantitatively a potential "active" effect of the forest (i.e., water purification per se) from the "passive" one being directly linked to the pressure degree of each LULC. (ii) The advantage of capturing the relationships between several water quality variables and forest cover implies that: (a) water quality variables with less frequent measurement during the studied decade could not be studied as they would have drastically reduced the number of observations; and (b) conclusions remain rather general and do not allow to discuss one particular variable in detail. However, we believe this approach has the following strengths: (i) it is based on public monitoring network data (several annual measurements for the processed pollutants) linked to the WFD and monitored in many other countries; (ii) as we did previously (Brogna et al., 2017b), we based this analysis on "real-life catchments" making conclusions more complex to draw but allowing for studying this phenomena at a regional scale and provide land planners with insights potentially contributing to a more sustainable resource management; and (iii) we applied a straightforward statistical approach, easier to apply and more efficient than physically based hydrologic/water quality models when observed data are limited in time but when datasets cover many different catchments. Furthermore, this statistical approach allows quantifying the effect of land cover on several pollutants concomitantly while controlling for autocorrelation and ecological factors, allowing for comparison between them. Finally, the statistical method was implemented in open source statistical software (R), which eases the replicability in other regions and/or through time.

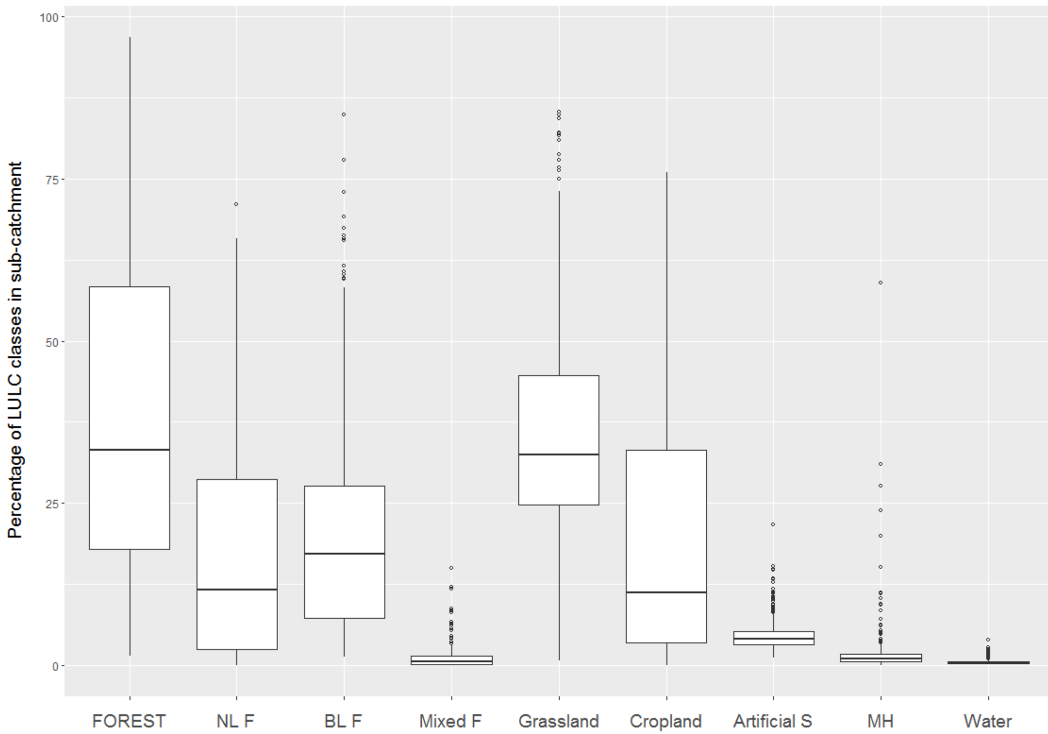
4.5 Conclusions

Our study demonstrates significant effects of forest cover on water quality, disentangles independent and shared effects of correlated LULC categories while controlling for autocorrelation, and applies a method to mine large monitoring datasets. Capturing effects of land cover on several water quality variables at the same time from measured data allows for comparison between them. This contributes to validation and refining of the hypothesis that forests improve water quality.

Further research could focus on measuring direct LULC change impacts on water quality in study areas where more detailed and regularly updated land cover datasets are available. In addition, spatial information on land use practices could be further integrated into the analysis to enrich the interpretation. LULC effects can also be studied in more detail by taking into account spatial heterogeneity through landscape metrics studies, as some authors did (Amiri and Nakane, 2008; Clément et al., 2017; Łowicki, 2012), or, e.g., focusing on specific locations or types of forest such as riparian forests. Study of residuals and outliers could bring to light how catchment management can mitigate effects on water quality, as there are sub-catchments where water quality is unexpectedly high or low based on its LULC profile. Finally, similar analyses could be performed with biological water quality data, such as with macroinvertebrate (Miserendino and Pizzolon, 2004) or diatom indices, which would bring complementary information.

The approach presented here, replicable in time and space, has a large application potential. First, it uses the publicly funded standard monitoring data linked to the WFD. Second, the analysis is based on "real-life" sub-catchments reflecting LULC heterogeneity and based on WFD water bodies, providing land planners and decision makers with directly applicable insights. Finally, we applied a straightforward statistical analysis, which are simpler, easier to apply, and more efficient than physically-based hydrologic/water quality models.

Supplementary Materials:



LULC percentages in the sub-catchments with NL F needle-leaved forest, BL F: broad-leaved forest, S: Surfaces and MH: shrubs - heathlands and moorlands

Chapter 5 Forest cover impact on instream water supply in terms of biological quality

This chapter is a submitted version of the following paper:

Brogna D., Dufrêne M., Michez A., Latli A., Jacobs S., Vincke C., Dendoncker N., (2017, submitted). Forest cover ensures good biological and physico-chemical water quality. Insights and nuances from a regional study (Wallonia, Belgium). Submitted to Journal of Environmental Management.

Preamble

This research main objective is the **study of the impact of forest cover on instream water supply in terms of biological water quality**. In this study, biological water quality is described by two indices based on macroinvertebrates and diatoms communities. We quantify, at the regional scale and across five natural ecoregions, the effect of forest cover on water quality at the riparian and catchment scales. We assess this effect while controlling for spatial, local morphology and population pressure variations and we quantify independent and shared effects between forest cover and the physico-chemical water quality, anthropogenic pressures (agriculture and population density) and local morphology.

Transversal methodological objectives are in here reached, through the development of a replicable approach (i) based on easily accessible data, monitored in many countries (i.e. EU Water Framework Directive monitoring datasets), (ii) using multivariate statistical methods and (iii) with main processes run in open source statistical software. The enlargement of scope of the derived results is reached through the study of 171 headwater monitoring stations spread across the region whose upstream catchments have mixed land covers.

Abstract

Forested catchments are generally assumed to provide higher quality water in opposition to agricultural and urban catchments. However, this should be tested in various ecological contexts and through the study of multiple variables describing water quality. Indeed, interactions between ecological variables, multiple land use and land cover (LULC) types, and water quality variables render this relationship highly complex. Furthermore, the question of the scale at which land use within stream catchments most influences stream water quality and ecosystem health remains only partially answered. This paper quantifies, at the regional scale and across five natural ecoregions of Wallonia (Belgium), the forest cover effect on biological water quality indices (based on diatoms and macroinvertebrates) at the riparian and catchment scales. Main results show that forest cover – considered alone – explains around one third of the biological water quality at the regional scale and from 15 to 70% depending on the ecoregion studied. Forest cover is systematically positively correlated with higher biological water quality. When removing spatial, local morphological variations, or population density effect, forest cover still accounts for over 10% of the total biological water quality variation. Partitioning variance shows that physico-chemical water quality is one of the main drivers of biological water quality and that anthropogenic pressures often explain an important part of it (shared or not with forest cover). The proportion of forest cover in each catchment at the regional scale and across all ecoregions but the Loam region is more positively correlated with high water quality than when considering the proportion of forest cover in the riparian zones only. This suggests that catchment-wide impacts and *a fortiori* catchment-wide protection measures are the main drivers of river ecological water quality. However, distinctive results from the agricultural and highly human impacted Loam region show that riparian forests are positively linked to water quality and should therefore be preserved.

5.1 Introduction

5.1.1 Freshwaters and water quality

Despite its crucial importance for the life of all beings (Haddadin, 2001; UN-Water, 2014), water and freshwater systems in particular are directly threatened by human activities (Loh et al., 2005; Meybeck, 2003; Millennium Ecosystem Assessment, 2005a; Vörösmarty et al., 2010). In response to global degradation of ecosystems and their services, water quality management is at the core of policies such as the US Clean Water Act (1972) and the European Water Framework Directive (Directive, 2000/60/CE) (European Commission, 2000). Water quality can be described by a huge number of variables which can broadly be classified into physical, chemical and biological categories (Boyd, 2015; Chapman, 1992). These groups of variables provide complementary information and are inter-related, but biological indicators have the advantage to assimilate long-term disturbance and stress trends in freshwater ecosystems while avoiding the complexity, costliness and high temporal variability linked to physico-chemical measurements (Allan, 2004; Bere and Tundisi, 2010; Giorgio et al., 2016). Among biological indicators, benthic macroinvertebrates are often used to determine the water quality notably because of their sensitivity to pollution, limited mobility, rapid response to external disturbance and dependence on the land environment around the stream (Mahler and Barber, 2017; Sharma and Rawat, 2009). Phytobenthos – of which diatoms are the main component – present a reduced mobility, a short generation time and a rapid response to environmental changes. Diatoms are tightly linked to physico-chemical changes. Being preserved in sediments, they are a good indicator of eutrophication, acidification and organic pollution (Delgado et al., 2012; Lobo et al., 2016). Therefore integrating information from diatoms and macroinvertebrates allows a better assessment of stream ecological integrity by bringing nuances in the responses to multiple pressures (Giorgio et al., 2016; Hering et al., 2006; Marzin et al., 2012; Soininen and Könönen, 2004).

5.1.2 Land use and Land cover impact on water quality

Land use and Land cover (LULC) are key landscape elements affecting water quality through their impact on non-point source pollution resulting from complex run-off and landscape interactions. Giri and Qiu (2016) stress the importance of assessing the relationship between LULC and water quality. According to them, improving the understanding of these relationships can help managing water quality in unmonitored watersheds but also providing recommendations to watershed managers and policymakers in the land planning decision process. Related to catchment and riparian degradation in particular, the question addressing the scale at which land use within stream catchments most influences stream water quality and ecosystem health remains only partially answered (Allan, 2004; Johnson et al., 1997; Sheldon et al., 2012; Sponseller et al., 2001). Several studies suggest that prevailing (Kail et al., 2012; Riva-Murray et al., 2002) and past (Harding et al., 1998) LULC characteristics of the whole stream catchments affect surface water quality. Other studies emphasise the impact of riparian LULC on water quality or stream habitat (Dosskey et al., 2010; Jackson et al., 2015). Finally, some studies compare scales of influence (i.e. catchment scale versus riparian scales), obtaining nuanced results on the land use effect on stream water quality according notably to the type of biological indicators and the ecological context of the sampling sites (Kosuth et al., 2010; Marzin et al., 2012, 2012; Sponseller et al., 2001). These studies show that assessing both scales of influence bring deeper insights when studying LULC impact on water quality (Vondracek et al., 2005).

Regarding the type of LULC, negative impact of agricultural intensification is reported in the literature (Stoate et al., 2001) mainly explained by the following processes: increased sedimentation, modified hydrological regimes, loss of high quality habitat, contamination from pesticides, increases in surface water nutriments (mainly N and P) (Allan, 2004; Herringshaw et al., 2011; Mahler and Barber, 2017). Urban land use – despite covering small areas – and urban intensification are also reported to negatively affect water quality (Kosuth et al., 2010; Riva-Murray et al., 2002). Forest, on the contrary, is usually associated with water containing less sediments and fewer nutriments (Neary et al., 2009; TEEB, 2010). Some studies showed impact of forest cover on instream water quality (Kosuth et al., 2010; Tong and Chen, 2002), on fish, macroinvertebrate and algal biomass (Stephenson and Morin, 2009). Specifically, forested riparian buffer zones are believed to have a positive impact on water quality through notably the reduction of the sediment load and nutrient concentrations in water (Dosskey et al., 2010; Fernandes et al., 2014; Naiman et al., 2005; Scarsbrook and Halliday, 1999). However, this is nuanced by studies explicitly assessing the effect of riparian forest compared to forest proportion in the whole catchment. For example, Stephenson and Morin (2009), in their study of the partial effects of forest cover on biomass and community structure metrics of algae, invertebrates and fish, never detected a significant partial effect of forest cover at the riparian scale. In conclusion, regarding LULC impact on biological water quality, literature shows general trends, especially opposing agricultural and urban LULC – associated with a negative effect on water quality – and forested land – broadly positively related with water quality, see e.g. Ding et al. (2013), Kosuth et al. (2010) or Theodoropoulos et al. (2015). However, issues of scales of influence and nuances brought by the type of studied biological indicators and the ecological context of study sites remain to be further explored. Also, to our knowledge and as observed by Tanaka et al. (2016), only few studies integrate information from macroinvertebrates, diatoms and physio-chemical water quality variables to get a broader picture of the forest cover impact on water quality.

5.1.3 Objectives

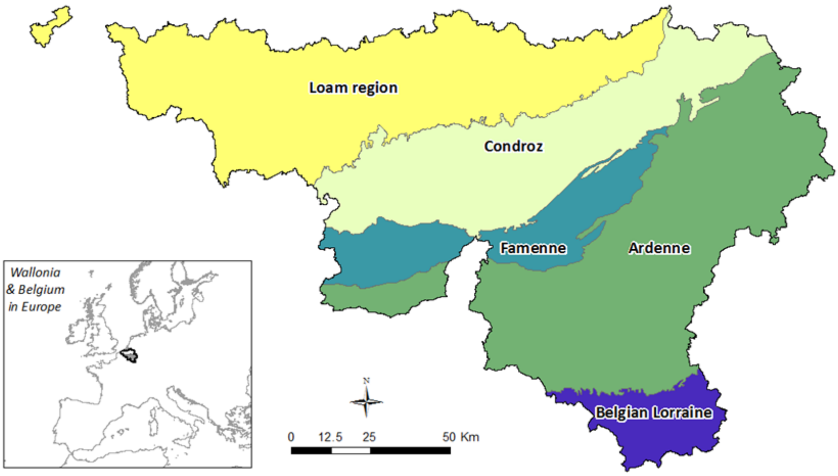
The main objective of this paper is, at the regional scale and across five natural ecoregions, to quantify the forest cover effect on biological water quality indices at the riparian and catchment scales. This objective is addressed through: (i) the assessment of this link while controlling for spatial, local morphology and population pressure variations, (ii) the quantification of independent and shared effects between forest cover and the physico-chemical water quality, anthropogenic pressures (agriculture and population density) and local morphology.

5.2 Material and methods

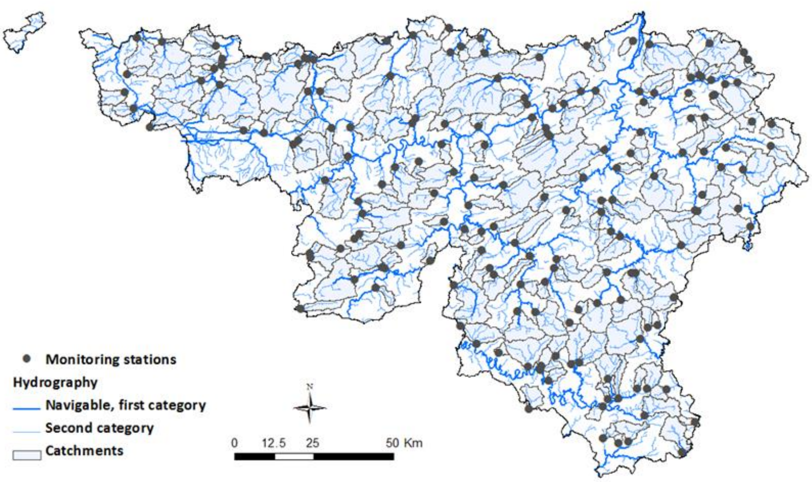
5.2.1 Study area

The study area is the southern region of Belgium (Wallonia) covering 16 898 km^2 (ca. 55% of Belgium's area, see Figure 5-1 A). We work on 173 headwaters stations located on the publically managed river network where biological and physico-chemical water quality data are monitored by the Walloon Public Service [WPS (SPW - DGO3, n.d.), Figure 5-1 B). These stations monitor headwater waterbodies and have non-overlapping upstream catchments (Figure 5-1 B &D). Figure 5-1 D shows forest cover distribution in waterbodies.

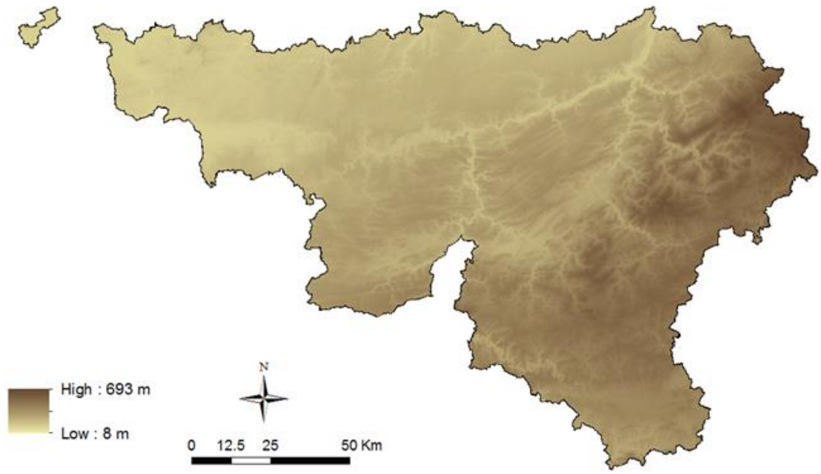
A. ECOREGIONS



B. MONITORING STATIONS – CATCHMENTS & HYDROGRAPHY



C. ELEVATION



D. FOREST COVER IN WATERBODIES – EU-WFD

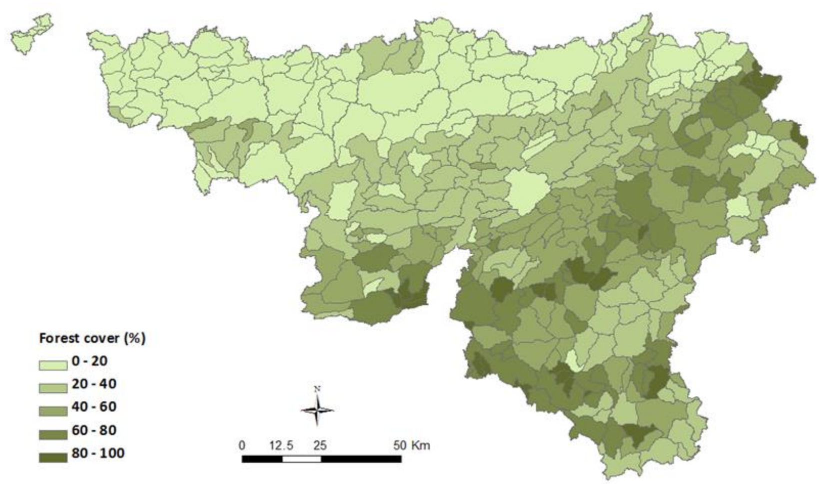


Figure 5-1. (A) Ecoregions in Wallonia; (B) Hydrography, water quality monitoring stations and corresponding catchments; (C) Elevation (source: regional LiDAR digital terrain model, http://geoportail.wallonie.be) and (D) Forest cover proportion (source: Top10VGIS) in waterbodies delineated within the EU Water Framework Directive

Wallonia presents relatively contrasted landscapes and can be divided into five natural ecoregions (Figure 5-1 A and Table 5-1). Noirfalise (1988) delineated these ecoregions according to pedological, botanical and agro-ecological criteria. Table 5-1 presents their main characteristics regarding LULC, topography, and rainfall distributions. Main ecological differences are found across an elevation gradient from the Loam to the Ardenne ecoregion. The Loam and the Condroz ecoregions located in lower elevation areas (Figure 5-1 C) mainly comprise agricultural and urban land uses, and present high human population densities. The Ardenne ecoregion mostly consists in forested and grassland landscapes with lower population density, but remains a highly managed region. The Famenne and the Belgian Lorraine ecoregions, bordering the Ardenne in the North and South respectively, present an intermediate context with an equal coverage of agricultural and forested land.

Table 5-1: Main ecological characteristics of Wallonia and its ecoregions [adapted from Michez et al. (2017); LULC source: Top10VGIS dataset].

	Area (km²)	**Rainfall** (mm / year)	**Mean altitude** (m)	**Mean slope** (%)	**Agr.** (%)	**Urb.** (%)	**For.** (%)	**Wat.** (%)	**Mean pop density**
Loam region	5192	825	103	4.8	68.8	19.2	10.3	<1	320
Condroz	3570	956	214	9.8	54.6	18.7	24.5	1.2	344
Famenne	1574	898	227	9.3	47.9	9.0	41.4	1.0	74
Ardenne	5710	1140	425	11	34.2	7.1	56.3	<1	44
Belgian Lorraine	851	934	322	9.1	46.3	10.3	41.6	<1	107
Wallonia	**16898**	**971**	**258**	**8.5**	**51.0**	**13.6**	**33.3**	**<1**	**208**

Agriculture in Wallonia is generally intensive with a negative impact on water quality through the use of mineral fertilizer, in particular nitrogen (N) and phosphorus (P), causing eutrophication and drinking water quality degradation. Even if declining since 1990, inputs of nitrogen and phosphorus were still above the European average in 2001 (SPW-DGO3-Direction de l'Etat Environnemental, 2014, 2007). Nitrogen still exceeded (about double) the European average in 2012 while phosphorus decreased to around half of the European average. Agricultural land is relatively heterogeneous across Wallonia and mainly consists of cropland and grassland. Their spatial distribution is relatively heterogeneous. In the Loam region, most agricultural lands (three quarters) are intensive cropland whereas in the

Ardenne, Belgian Lorraine and Famenne ecoregions, most agricultural lands are grassland (up to 85% in the Ardenne). In the Condroz ecoregion, grassland and cropland share comparable areas.

Most of the forest cover in Wallonia is represented by either needle-leaved (44%) or broad-leaved forests (53%), the rest being classified as mixed forest (3%) (source: Top10VGIS). Needle-leaved forests – mainly located in the Ardenne – are intensively managed with the use of exotic species (mainly spruce (*Picea abies*) but also Douglas fir (*Pseudotsuga menziesii*), larches (*Larix* sp.), and pines (*Pinus sylvestris* and *P. nigra*)). These are conducted in even-aged stands with systematically clear-cuttings, and drainage infrastructure when located on wet soils. Broad-leaved forests – which, in contrast with needle-leaved forests, spread across Wallonia – are largely dominated by oaks (*Quercus robur* and *Q. petraea*) and beech (*Fagus sylvatica*). Other species such as birch (*Betula pendula*), ash (*Fraxinus excelsior*), maple (*Acer pseudoplatanus*), and hornbeam (*Carpinus betulus*) are also present (Alderweireld et al., 2015).

5.2.2 Datasets

The variables used in this study and the datasets on which they are based are provided in Table 5-2. These variables are either response variables (biological water quality indices) or explanatory variables linked to LULC, physico-chemical water quality, anthropogenic pressures in upstream catchment, local morphology or elevation. These variables are described in sections 5.2.2.1 to 5.2.2.3.

Table 5-2. Response variables (i.e. biological water quality indices), explanatory variables used in this study and the basis datasets & source

TYPE OF VARIABLE	VARIABLE		UNIT	DATASET /SOURCE
Response variables				
Biological indices	Standardized Global Biological Index/reference value for the corresponding river type Ratio	IBGN-R	-	WPS – EU-WFD monitoring
	Specific Polluosensitivity Index/reference value for the corresponding river type Ratio	IPS-R	-	
Explanatory variables				
LULC in upstream spatial unit	Forest, grassland/cropland cover		(%)	TOP10VGIS (NGI)
Physico-chemical water quality	Dissolved Oxygen	DO	(mgO_2/l)	WPS – EU-WFD monitoring
	Dissolved Organic Carbon	DOC	(mgC/l)	
	Total phosphorus	TP	(mgP/l)	
	Ammonium	NH_4	(mgN/l)	
	Nitrites	NO_2		
	Nitrates	NO_3		

TYPE OF VARIABLE	VARIABLE		UNIT	DATASET /SOURCE
Physico-chemical water quality	pH	pH	-	WPS – EU-WFD monitoring
	Chloride	Cl	(mg/l)	
	Sulphate	SO_4	(mg/l)	
	Suspended materials	SusMat	(mg/l)	
Anthropogenic pressures in upstream catchment	Population density		(Hab/km^2)	Stat Bel
	Agricultural cover		(%)	TOP10VGIS (NGI)
Local morphology	Channel width	CW	(m)	Extracted from WPS LiDAR digital terrain model
	Emerged channel depth	ECD	(m)	
	Local sinuosity of the river sector	sin	(%)	
	Catchment Area	Area	(km^2)	
Elevation	Average elevation of the upstream catchment	Elev	(m)	WPS LiDAR digital terrain model

5.2.2.1 Biological and physico-chemical water quality

Biological and physico-chemical water quality are described by variables measured as part of the monitoring of water bodies quality performed by the WPS for the EU-WFD (SPW-DGO3-Direction de l'Etat Environnemental, 2016). Dahm et al. (2013) highlight the potential of broad datasets such as EU-member states 'water quality monitoring data and argue that those represent the European water bodies much better than restricted datasets from local studies and projects. Processing these datasets with appropriate methods offers an opportunity to study LULC impact on ecological integrity at different scales and combining various indicators types. We selected six years of data (2009-2014) corresponding to the last EU-WFD cycle (data from 2015 were not validated yet). Biological water quality is described through annual values of the macroinvertebrates index and the diatoms index. The macroinvertebrates index is based on the French IGBN (i.e. "Standardized Global Biological Index") that was adapted to Wallonia (Vanden Bossche, 2005). The IBGN score, with a range from 0 (no indicator taxa) to 20, is obtained by crossing two sub-indices: the "faunal indicator group" reflecting pollution sensitivity and the taxonomic diversity class reflecting habitat quality. The index based on benthic diatoms is the IPS ["Specific Polluosensitivity Index", see Coste in CEMAGREF (1982)].

We selected all stations that monitor a headwater waterbody. The resulting dataset is composed of 319 measurements related to 173 stations spread across Wallonia (Figure 5-1 B and D). Stations represent different kinds of "control" type with regard to the EU-WFD. Indeed, stations are almost equally divided into "additional" and "operational" control corresponding to relatively 'good state' waterbodies and impacted waterbodies respectively. Surveillance station are also part of the dataset. Both biological water quality indices were divided by the reference value for the corresponding river type (SPW-DGO3-Direction de l'Etat Environnemental, 2016) to obtain comparable indices across the region (Kosuth et al., 2010). The obtained indices are further referred to as IBGN-R and IPS-R. The physico-chemical water quality is described by annual average values of the following variables: Dissolved Oxygen, Nitrates, Chloride, Sulfates, pH, Temperature, Total Phosphorus, Nitrites, Ammonium, Dissolved Organic Carbon and Suspended Materials. We applied Log- or square-based transformations when needed to improve normality of variables' distribution.

5.2.2.2 Land use and land cover data and pressures

We used the Top10VGIS land cover data set version of 2010 from the Belgian National Geographic Institute (NGI, www.ngi.be) to characterize the land cover influencing the water quality at the monitored station. This vector data set, which covers the whole of Belgium, is based on the NGI topogeographic data that classifies LULC into 37 classes. In this study, we reclassified it in six classes of interest by either keeping the original land cover classes as such or grouping them. The classes of interest are forest (i.e. needle-leaved, broad-leaved and mixed forest), cropland and grassland further grouped into agricultural land, artificial surfaces, water surfaces and shrubs-heathlands. We assumed as in other studies (Brogna et al., 2017a, 2017b) that the evolution of the retained classes in the region was minor throughout the studied period (2009-2014).

To relate the LULC to water quality, we intersected the Top10VGIS dataset with three distinct related upstream spatial units: riparian buffer, outside this buffer, and the whole catchment. To our knowledge in most of studies, authors use a fixed-distance buffer to study riparian LULC impact on water quality [e.g. (Boyer-Rechlin et al., 2016; de Oliveira et al., 2016; Marzin et al., 2013; Sliva and Dudley Williams, 2001; Sponseller et al., 2001). In this study, we treat channels of relatively contrasted morphology even within the same ecoregion and range of catchment sizes, and a similar buffer width might or might not represent the same riparian zone extent according to the riparian topography and the parameters of the associated river (channel size, hydrological regime). Hence, we based our definition of the riparian area on a regional geographic layer representing areas subject to flooding by overflowing for return periods of 25, 50 and 100 years (see *"aléa d'inondation"*, http://geoportail.wallonie.be). We selected the spatial area corresponding to the 100 years flooding which is a rather large delineation of the riparian zone. We believe this choice renders our riparian zone definition closer to its hydromorphological reality.

Regarding the catchment scale, we automatically extracted upstream catchments from a regional LiDAR digital terrain model (1 m GSD) provided by the WPS (see Figure 5-1 C, http://geoportail.wallonie.be). The most represented LULC classes in our dataset (see section 5.2.1) are forest and agricultural cover (grassland and/or cropland cover) (see boxplots in

Supplementary Materials Figures S1). The Loam region presents the lowest forest cover proportion in upstream catchments.

We computed population density in each upstream catchment based on a statistical administrative Belgian database from 2008 (Statistics Belgium, n.d.) to complement agriculture proportion and create a proxy matrix for anthropogenic pressure. Population density was computed for each statistic sector scale (smallest administrative spatial unit where population data are available). Then, density values were linked to sectors centroids to derive a spatial grid that was used to extract median density values for each upstream catchment.

5.2.2.3 Physical characteristics of stations and catchments

We used the regional LiDAR digital terrain model to compute average elevation over the catchments. We then computed three main local morphological parameters of the river network following the approach of Michez et al. (2017). We extracted from the same LIDAR digital terrain model and for every monitoring station, the channel width (m) and the emerged channel depth (m) associated to the corresponding river reach. We also computed the local sinuosity (%) of the upstream river sector associated to each station.

5.2.3 **Spatial scales of analysis**

Every analysis in the study was run over six different extents: at the regional (Wallonia) study scale and within each of the five ecoregions. This allows providing a general picture for the region and to analyze trends and differences across ecoregions. Regarding the forest cover explanation power of biological water quality, we tested it on two distinct upstream spatial units within each catchment: the functional riparian buffer and outside this buffer (Figure 5-2 left and centre). For the first analysis, we also performed analysis on the percentage of forest over the whole catchment (Figure 5-2, right) – third upstream spatial unit (section 5.2.4.1) to compare it with the percentage of forest outside the functional riparian buffer.

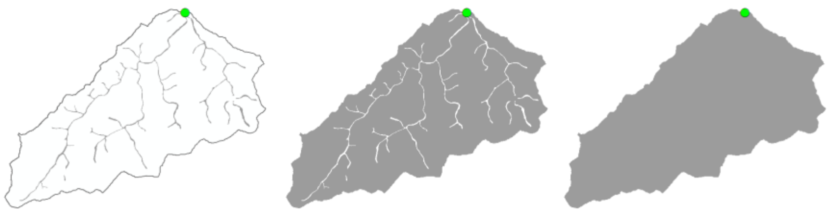


Figure 5-2. Upstream spatial units (grey). Left : functional riparian buffer; centre: outside the functional riparian buffer and right : whole catchment.

5.2.4 Forest cover link with biological water quality

We ran statistical multivariate analysis to fulfil the objectives of this study and exploit the potential of broad datasets such as EU-member states 'water quality monitoring. These are described in sections 5.2.4.1 to 5.2.4.3.

5.2.4.1 Functional riparian buffer or catchment scale?

We performed redundancy analysis [RDA, see Legendre and Legendre (2012c), R package vegan (Oksanen et al., 2017)] for the six extents and for the three upstream spatial units where forest cover was computed: functional riparian buffer, outside this buffer, and the whole catchment. Redundancy analysis is a multivariate analysis that allows capturing the linear relationship between several dependent variables and one or several explanatory variables. In this case, RDA quantifies the percentage of biological water quality variability explained by forest cover proportion. RDA also allows quantifying and excluding the variability explained by other covariates. We ran these RDA on centered and scaled variables because of the heterogeneity of the variables units.

Bio WQ ~ Forest (Equation type 1)

where Bio WQ = matrix of biological water quality indices (i.e. IBGN-R and IPS-R), and Forest = percentage of forest cover in the upstream spatial unit.

Following results interpretation, complementary RDA will be ran to quantify the explanation power of main LULC types' proportions inside and outside the functional riparian buffer on biological water quality.

5.2.4.2 Forest cover explanation power when controlling for spatial autocorrelation, local morphology and population pressure

We tested the impact of several variables or group of variables on the forest cover explanation power by putting them as covariate in RDA's. As in Brogna et al. (2017a), we tested the effect of elevation as a mean of controlling spatial autocorrelation (Equation type 2, with covariate being average elevation of the upstream catchment). We present these results for the six extents even though this is especially true at the Walloon regional scale. Indeed, a strong continuous ecological gradient exists in Belgium and is highly correlated to elevation (Dufrene and Legendre, 1991; A Noirfalise, 1988). Dufrêne and Legendre (1991) showed that elevation, although not exceeding 700 m in Belgium, explains almost all the geographic structure of several ecological variables given their spatial autocorrelation.

Bio WQ ~ Forest + Condition(*Covariate*) (Equation type 2)

where Bio WQ = matrix of biological water quality indices (i.e. IBGN-R and IPS-R), Forest = percentage of forest cover in the upstream spatial unit, and 'Covariate' = variable whose effect on Bio WQ is removed before quantifying the forest cover effect.

We also tested the effect of local morphology by putting the following variables as covariates (Equation type 2): sinuosity of the river sector associated to the station, local channel width, emerged channel depth and upstream catchment area. Following interpretation of these results, significant covariates will be kept for further analysis (section 5.2.4.3). Finally, we tested the effect of population density in the same way.

These tests allow deepening the interpretation of the forest cover link even effect on water quality.

5.2.4.3 Forest cover: Independent and shared explanation power with anthropogenic pressures and physico-chemical water quality

We ran variation partitioning (Legendre and Legendre, 2012c) to quantify independent and shared forest cover explanation power with physico-chemical water quality, anthropogenic pressures and other potentially relevant covariates from the analysis described in section 5.2.4.2. We computed adjusted redundancy statistics R^2 to provide unbiased estimates of the explained fractions of variance (Peres-Neto et al., 2006). Anthropogenic pressures are represented by the proportion of agricultural land and the population density in the upstream catchment. Given the high correlation between physico-chemical water quality variables, we reduced information by selecting, for each extent of study, the two variables most post-correlated with the first and second axes of a PCA on biological water quality, respectively. Results of variation partitioning are only presented for variables linked to significant individual and partial RDA models (i.e. p value < 0.05). Furthermore, we illustrated the link between explanatory variables (physico-chemical water quality, anthropogenic pressures and local morphology variables) and biological water quality through Principal Component Analysis (PCA) on biological variables and post-correlations.

5.3 Results

5.3.1 Functional riparian buffer or catchment scale?

Figure 5-3 shows differences in forest cover impact on biological water quality according to the three distinct upstream spatial units were forest cover is computed. Figures represent the proportion of variability in the biological water quality dataset explained by forest cover proportion. We do not present the Famenne ecoregion results, as the models were not significant (evaluation with permutation tests) (see further details in Table 5-4). Forest cover explains around a third of the biological water quality variability in Wallonia. The forest cover link with biological water quality is far more demonstrated in the Belgian Lorraine (around 70% of explained variability). Figure 5-3 also illustrates that the proportions of variability explained by proportion of forest cover in the whole catchment and in the area outside the functional riparian buffer are highly similar. Hence, to ease the reading and work on spatially independent areas (i.e. non overlapping areas), we will only keep the following upstream spatial units for further analyses: inside the functional riparian buffer and outside it.

In Wallonia and for every ecoregion but the Loam region, the forest cover proportion in the area outside the functional riparian buffer slightly better explains the biological water quality than the proportion of forest cover in the functional riparian buffer.

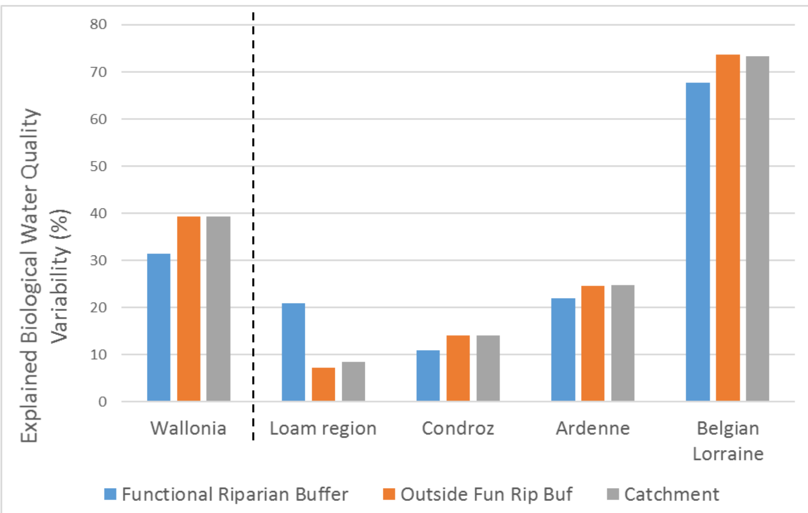


Figure 5-3. Biological water quality variability explained (redundancy analysis) by forest cover proportion in three upstream spatial units: Functional riparian buffer, outside this buffer (Outside Fun Rip Buf) and in the whole catchment. Results from significant models for Wallonia scale and in ecoregions.

The Loam region shows distinctive results compared to the other ecoregions. Therefore, we provide complementary analyses results in Table 5-3 to refine the interpretation of the forest cover link with biological water quality in this ecoregion. This table presents details of RDA results quantifying biological water quality variability explained by forest, cropland and grassland cover respectively. We distinguished between cropland and grassland as their distribution is variable according to the upstream spatial unit considered (see

Figure S1 B in supplementary materials). Results compare riparian forest cover and forest cover outside riparian buffer. F value and significance of the models from permutation tests are also provided.

*Table 5-3. Redundancy analysis results for the Loam region: biological water quality (IBGN-R and IPS-R) variability (%) explained by proportion of forest, cropland and grassland cover in functional riparian buffer and outside this buffer. F value and models significance from permutation tests (correspondence with p value s: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1)*

Scale where proportion of LULC	LULC	Variability explained (%)	F	Model Significance
Functional Riparian Buffer	Forest	21	20	***
	Cropland	11	9	**
	Grassland	4	3	.
Outside Functional Riparian Buffer	Forest	7	6	**
	Cropland	3	2	Non sign
	Grassland	4	3	.

Forest cover proportion inside the riparian buffer explains around 21% of the biological water quality variability in the Loam region with a highly significant model (p value of 0.001). Cropland explain 11% while grassland cover explains only 4% with a barely significant model (p value between 0.05 and 0.1). Regarding LULC outside the functional riparian buffer, results show that forest cover explains 7% of the water quality variability and is the only significant model. Indeed, cropland-based model is non significant and the grassland one is again at the limit of model significance.

5.3.2 Forest cover explanation power when controlling for spatial autocorrelation, local morphology and population pressure

Details of RDA results quantifying forest cover explanation power of biological water quality with or without covariate for the six extents are presented in Table 5-4. Results compare functional riparian forest cover and forest cover outside this buffer link with biological water quality. F value and significance of the models from permutation tests are also provided. Figure 5-4 presents the global trend, found in every model, through the regional forest cover – computed outside the riparian buffer – link with biological water quality when controlling for spatial autocorrelation through elevation. The factorial plan is constituted by the constrained axis (RDA1) and the first residual axis (PC1).

*Table 5-4. Redundancy analysis results: biological water quality (IBGN-R and IPS-R) variability (%) explained by proportion of forest cover in functional riparian buffer and outside this buffer respectively. Results for Wallonia and ecoregions. Variability explained without removing any effect (Equation 1) and when removing Elevation, Local morphology or Population density effects (Equation type 2). F value and models significance from permutation tests (correspondence with p values: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1)*

				No Covariate			Covariate : Elevation			Covariate : Local Morphology			Covariate : Population density		
Region	**n stations**	**n obs**	**Scale where proportion of forest**	**Variability explained (%)**	**F**	**Model Significance**	**Variability explained (%)**	**F**	**Model Significance**	**Variability explained (%)**	**F**	**Model Significance**	**Variability explained (%)**	**F**	**Model Significance**
Wallonia	173	319	Functional Riparian Buffer	29.2	131	***	11.7	66	***	28.9	138	***	19.0	91	***
			Outside Fun Rip Buf	38.1	195	***	13.0	75	***	36.8	200	***	26.9	146	***
Loam region	44	79	Functional Riparian Buffer	20.9	20	***	18.6	18	***	12.7	12	***	16.4	18	***
			Outside Fun Rip Buf	7.3	6	**	8.0	7	**	7.3	7	**	7.9	8	**
Ardenne	62	123	Functional Riparian Buffer	20.5	31	***	23.7	38	***	17.1	30	***	14.6	23	***
			Outside Fun Rip Buf	23.6	37	***	26.7	44	***	19.8	36	***	17.3	29	***
Condroz	39	71	Functional Riparian Buffer	8.6	7	*	6.7	5	*	14.8	13	***	6.0	6	*
			Outside Fun Rip Buf	10.9	8	**	8.7	7	**	14.3	13	***	8.9	9	**
Belgian Lorraine	10	15	Functional Riparian Buffer	67.7	27	***	60.3	23	***	53.7	35	**	42.0	22	***
			Outside Fun Rip Buf	73.7	36	***	65.2	30	***	54.6	39	***	43.3	24	***
Famenne	18	31	Functional Riparian Buffer	4.4	1	NON SIGN	3.4	1	NON SIGN	2.0	1	NON SIGN	2.3	1	0.452
			Outside Fun Rip Buf	5.6	2	NON SIGN	7.7	2	NON SIGN	1.9	1	NON SIGN	0.9	0	0.759

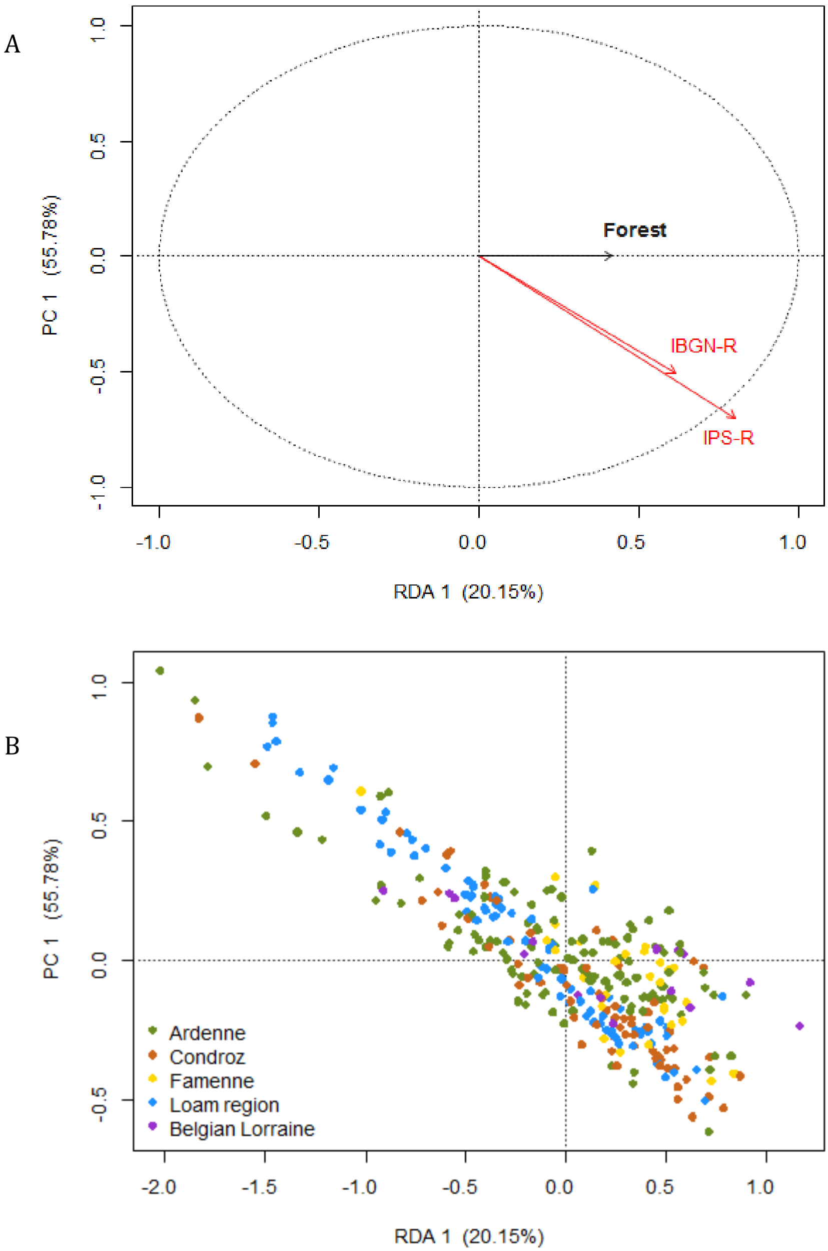


Figure 5-4: RDA results showing the link between forest cover and biological water quality indices (IBGN-R and IPS-R). X-axis represents the constrained axis and y-axis, the first residual component. (A) Variables correlation circle plot, (B) Individuals plot sorted per ecoregion.

Forest cover is systematically related to higher biological water quality whether for diatoms index (IPS-R) or macroinvertebrates index (IBGN-R) (see e.g. Figure 5-4). When controlling for spatial autocorrelation through the elevation factor at the Walloon scale, the biological water quality variability explained by forest cover drops from 31 to 13% and from 39 to 14% if computed in the functional riparian buffer or outside this buffer respectively. Elevation effect is, as expected, less important within the ecoregions as these are more homogeneous in terms of elevation and ecological factors. The local morphology impact on forest cover explanation power of biological water quality is small or even negligible at the Walloon scale and for the Ardenne and Condroz ecoregions. For the latter, removing morphological effect even increases the proportion of variability explained by forest cover. This increase is due to the fact that removing covariates effect might also remove part of the residual variability, hence enhancing the proportion of variability explained by the active variable. The situation is different in the Belgian Lorraine where removing local morphology effect decreases the proportion of variability explained by forest cover by 14 and 19 % when forest cover is computed in the functional riparian buffer or outside this buffer respectively. Despite this, Belgian Lorraine remains the ecoregion where forest cover best explains biological water quality. Population density effect on the relationships between forest cover and biological water quality is at every extent relatively important. Indeed, it reduces the proportion explained by forest cover (outside the functional riparian buffer) from around one third of its importance at the Wallonia scale, in the Ardenne and Belgian Lorraine and slightly less in the Condroz. Model in the Famenne ecoregion are non-significant.

5.3.3 Forest cover: Independent and shared explanation power with anthropogenic pressures and physico-chemical water quality

This subsection presents results from variation partitioning computed to isolate independent and shared explanatory power of the biological water quality between: (i) forest cover computed in or outside the functional riparian buffer according to the strength of the link with biological water quality (cf. Figure 5-3), (ii) anthropogenic pressures represented by population density and agricultural proportion in the catchment and (iii) physico-chemical water quality. Following the analyses presented in previous sections, we added local morphology variables when relevant (i.e. for the Loam region and Belgian Lorraine region). Derived plots are presented in Figure 5-5. Figures inside each subspace are positively adjusted coefficients of determination (expressed in percentage) and represent the variability explained by each subspace. Results for every study scale are presented except for the Famenne ecoregion where models are not significant. Biological water quality dataset PCA biplots with – as supplementary variables –each potential variable in this variation partitioning analysis are provided in supplementary materials (Figure S2).

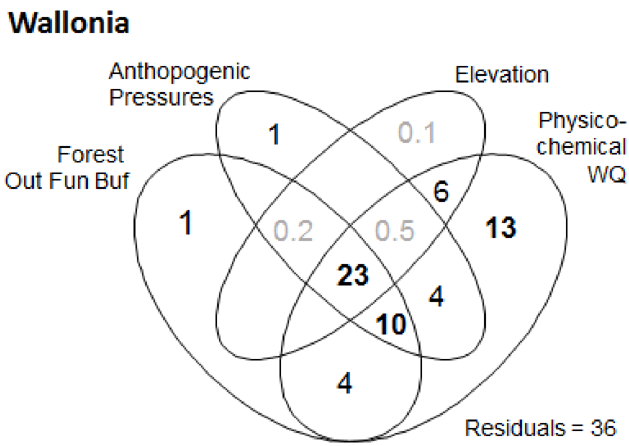
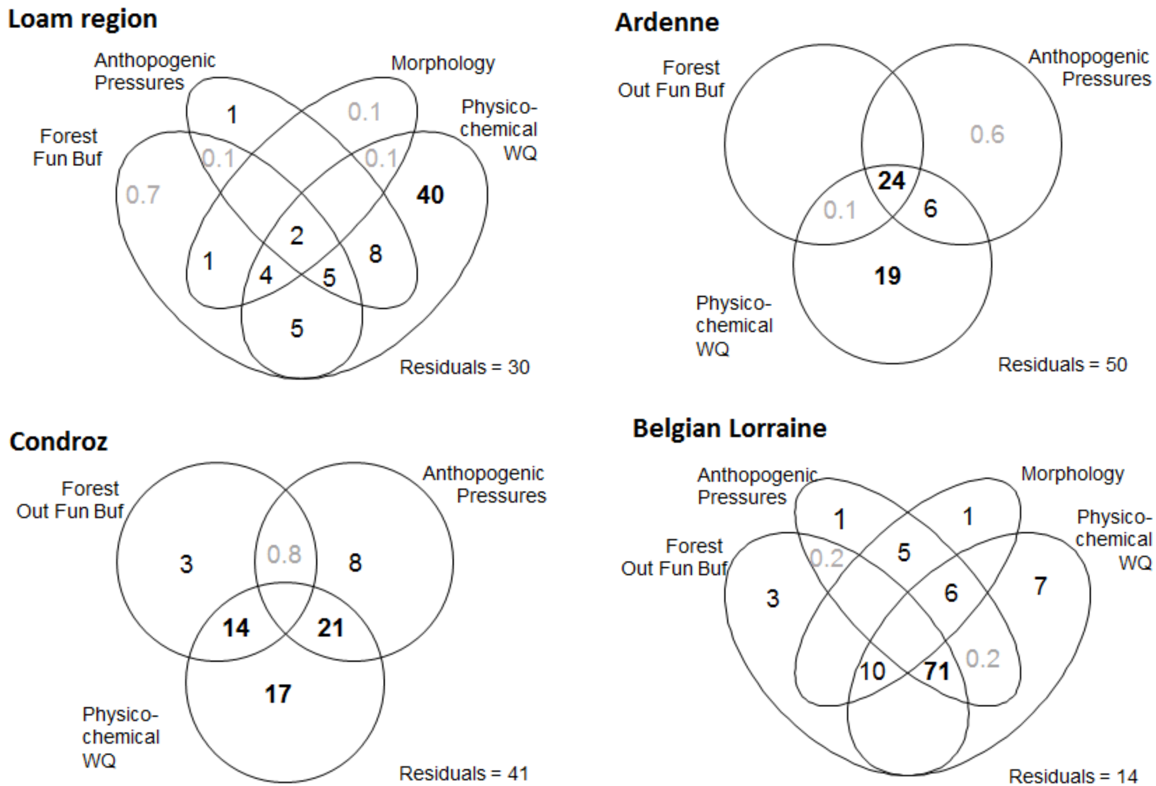


Figure 5-5 (here Wallonia, next page, ecoregions). Venn diagrams of the biological water quality variance partitioning into proportion of forest cover in the functional riparian buffer (Forest Fun buf) or outside the functional riparian buffer (Forest Out Fun Buf), physico-chemical water quality (PC), local morphology variables and anthropogenic pressures: proportion of agricultural land and population density in the catchment. Figures are positively adjusted coefficients of determination (expressed in percentage) and represent the variability explained by each subspace being either a single variable or shared effect between two or more variables


Loam region
Anthopogenic Pressures
Morphology
Physico-chemical WQ
Forest Fun Buf
1
0.1
0.1
0.1
0.7
40
2
8
1
4
5
5
Residuals = 30
Ardenne
Forest Out Fun Buf
Anthopogenic Pressures
0.6
24
0.1
6
19
Physico-chemical WQ
Residuals = 50
Condroz
Forest Out Fun Buf
Anthopogenic Pressures
3
0.8
8
14
21
17
Physico-chemical WQ
Residuals = 41
Belgian Lorraine
Anthopogenic Pressures
Morphology
Physico-chemical WQ
Forest Out Fun Buf
1
1
0.2
5
3
6
7
10
71
0.2
Residuals = 14

All models explain a relatively high proportion of biological water quality variability revealing that most factors driving water quality (or correlated to them) are considered. Models for the Belgian Lorraine and Loam region better explain biological water quality variability (13 and 29% residuals respectively) whereas residuals are higher at the regional scale (36%) and for the Condroz or Ardenne regions (41% and 50% respectively).

As shown on Figure 5-5, physico-chemical water quality explains on its own relatively high proportions of biological water quality variability: 40% in the Loam region, 19% in the Ardenne, 17% in the Condroz and 13% at the regional scale.

PCA analysis (see details in supplementary materials, Figure S2) shows that both IBGN-R and IPS-R are systematically opposed on the first PC to total phosphorus, ammonium, sulfates, nitrites, suspended materials, chloride, dissolved organic carbon, water temperature, and anthropogenic pressures. This is true in every extent specific case except for the Loam region and the Famenne where agricultural cover is not correlated with this first axe while population density is negatively correlated with high water quality indices values. Analysis across this first PCA component also shows that high biological water quality is systematically positively correlated with dissolved oxygen and forest cover proportion inside or outside the functional riparian buffer. Nitrates in most of situations are correlated with low biological water quality except for the Loam region and – to a lesser extent – for the Condroz ecoregion. The second axis, which represents the differences between diatoms and macroinvertebrates, explains far less than the first component.

Forest cover explanation of biological water quality is often shared (i.e. is inseparable) with physico-chemical water quality and anthropogenic pressures (Figure 5-5). This is especially true in Wallonia, Ardenne and Belgian Lorraine whereas this effect is in some cases proven to be relatively independent from anthropogenic pressures such as in the Condroz and Loam region.

5.4 Discussion

5.4.1 Preamble: forest cover in this study

In order to fulfil this study objectives, we chose to study the “forest cover” through a proportion of forest cover in upstream catchment or functional riparian buffer. Forest cover in this study includes various forests in terms of management, stand age, tree density, species combination, local conditions. Furthermore, studied catchments are what we can call “real-life” catchments with mixed land covers – with high variability that we exploit through statistical analyses – and various local conditions that we discuss and attempt to control in the same analyses (trough e.g. the ecoregion scale analysis). This renders results sometimes more difficult to interpret but also more linked to the landscape scale and therefore more connected to land planning.

5.4.2 Forest cover link with biological water quality

Several insights were derived while addressing the main objective of this paper being: at the regional scale and across five natural ecoregions, to quantify the forest cover effect on biological water quality indices at the catchment and functional riparian scales. First, forest cover is systematically related to higher biological water quality described by diatoms and macro-invertebrates community-based indexes. This is true no matter the extent of study. This is interesting as we study relatively heterogeneous LULC distributions, and in particular, regarding distributions of cropland and grassland covers. This finding corroborates studies associating forest cover with higher biological water quality (contrasting with agriculture and urban LULC) (Dahm et al., 2013; Ding et al., 2013; Kosuth et al., 2010; Theodoropoulos et al., 2015). However, comparing studies in detail is tricky as scales of LULC characterisation, selected biological indices and control variables are often study specific.

Regarding the scale at which forest cover within stream catchments most influences stream water quality, results vary according the extent of study. Main trend – i.e. in Wallonia and for every ecoregion but the Loam region – is that the forest cover proportion in the area outside the functional riparian buffer slightly better explains the biological water quality than the proportion of forest cover in the functional riparian buffer. This trend is in line with several studies highlighting that catchment-wide disturbances are the most influential determinants of river ecological quality (Allan, 2004; Clapcott et al., 2012; Dahm et al., 2013; Marzin et al., 2012, 2013; Stephenson and Morin, 2009).

Regarding the quantification and significance of the forest cover link with biological water quality, results show that forest cover – considered alone – explains around one third of the biological water quality at the regional scale and from 15 to 70% depending on the ecoregion studied. The Belgian Lorraine – were this link is the most demonstrated – is characterised by highly contrasted catchments and a relatively high biological water quality variation. Removing the influence of spatial autocorrelation and ecological factors decreases, at the Walloon scale, the biological water quality variability explained by forest cover from 31 to 13% and from 39 to 14% if computed in the functional riparian buffer or outside this buffer respectively. This result is similar with the quantitative assessment of forest cover effect on physico-chemical water quality in Wallonia (Brogna et al., 2017a). Intra-ecoregion effect of elevation is, as expected, less important within the ecoregions as these are more homogeneous in that respect. Local morphology impact on forest cover explanation power of biological water quality is diverse according to the extent of study. Indeed, it is small or even negligible at the Walloon scale and for the Ardenne and Condroz ecoregions whereas it is more important in the Belgian Lorraine. The important effect in this ecoregion can be explained by the highly contrasted morphological profiles of local rivers. Population density effect on the relationships between forest cover and biological water quality is at every extent relatively important, reducing the proportion explained by forest cover (outside functional riparian buffer) from around one third at the Wallonia scale, in the Ardenne and Belgian Lorraine and slightly less in the Condroz. Non-significance of any model in the Famenne ecoregion might be due to the complexity of water flows through the limestone subsoil.

Quantification of independent and shared explanation power of biological water quality between forest cover and the physico-chemical water quality, anthropogenic pressures (agriculture and population density) and local morphology reveals that physico-chemical water quality explains on its own relatively high proportions of biological water quality variability. This finding is in line with Dahm et al. (2013)'results identifying physico-chemical water quality as one of the main discriminating factor of biological water quality. This renders interpretation even more complex as physico-chemical water quality has been proven to be linked to LULC and forest in particular (Brogna et al., 2017a). This is also in line with the nutrient enrichment shown in European studies and in Wallonia [e.g. European Environment Agency (2012) or SPW-DGO3-Direction de l'Etat Environnemental (2014)]. Our study also highlights that forest cover explanation of biological water quality is often inseparable from physico-chemical water quality and anthropogenic pressures. However, it is in some cases interestingly proven to be relatively independent from anthropogenic pressures such as in the Condroz and Loam region. In the Loam region, complementary analysis shows that the proportion of agricultural land (i.e. grassland and cropland) at the catchment scale is not proven to have any link (RDA models are non-significant, $p>0.05$) with biological water quality whereas forest cover is at any spatial upstream unit scale. This finding is interesting in terms of land planning. Indeed, the fact that forests are mostly present in the functional riparian buffer in this ecoregion while relatively absent in the rest of the catchments combined with agricultural models being non-significant at the catchment scale let us believe that the computed forest link with biological water quality represents a "real" forest effect. Furthermore, study of the functional riparian buffer LULC explanation power of water quality revealed that only cropland cover model was significant (i.e. not grassland cover, $p>0.05$) and explained a twice-lower proportion of variability than forest cover. Non significance of grassland cover effect model could be linked to the existing diversity of management. Indeed, some grassland are enriched with Nitrogen, Phosphorus and Potassium while other are not. Cattle grazing pressure might also influence water quality through fine sediments transfer to the stream.

Consequently, in these catchments, riparian forests should be protected because of their positive effect on biological water quality. This is in line with Tran et al. findings (2010) showing a stronger correlation between LULC and stream water quality at the 200-m riparian buffer than that of the watershed. These authors also suggest that the presence of a riparian buffer zone between streams and agricultural and urban areas might reduce contamination from non-point source pollution. On the other hand, the fact that we did not detect this preponderance of a riparian effect in the other ecoregions and at the regional scale suggests, as Stephenson and Morin (2009) or Harding et al. (1998) noted in their study, that maintenance or preservation of habitat fragments in the riparian zone will not be sufficient to preserve ecological instream quality from catchment-wide impacts. Rather, this requires protection measures at the catchment scale.

5.4.3 Limitation, strengths of the study and perspectives

Some limitations of this study should be pointed out. First, the approach hardly allows for quantitatively isolating a potentially “active” effect of forest cover (i.e., water purification *per se*) from a “passive” one directly linked to the degree of pressure of other LULC on water quality. Then, we chose to describe biological water quality through integrated indices used in the waterbody quality assessment in the EU-WFD context. These indices are designed to be as comparable as possible between regions and are simplified indicators. This limits the analysis’ sharpness and using other information as biological traits could fine tune the analysis (Mondy and Usseglio-Polatera, 2013). In particular, as forest cover link with physico-chemical and biological water quality has been established in this study and in Brogna et al. (2017a), further studies could concentrate on biological traits in order to detail the forest cover impact on ecological processes of macroinvertebrates communities and eventually highlight trees species and management effects. However, we believe that the biological indices used in this study given their diversity (based on diatoms and macroinvertebrates communities) and their integrative character (of physical, chemical river quality status) are relevant to fulfil our study objectives.

We believe this study present the following strengths: (i) it is based on a relatively large public monitoring network data linked to the EU-WFD and thus monitored across Europe which makes it rather easily replicable in other European contexts, (ii) the database covers contrasted ‘real-life’ and heterogeneous catchments in Wallonia making the conclusions of this study more general, (iii) this study integrates physico-chemical and biological indices allowing to quantify the strong relationship between them, (iv) the different study extents (i.e. regional and sub-regional) allow to assess main regional trends and strengthen results while nuancing them according to local characteristics.

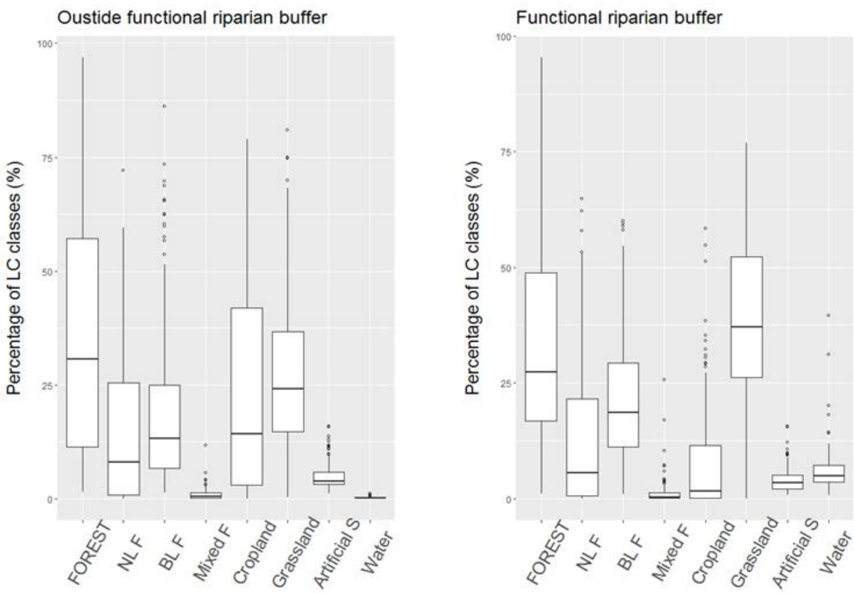
5.5 Conclusion

The main objective of this study was to quantify the forest cover link with biological water quality indices (macroinvertebrates and diatoms) at the riparian and catchment scales. This analysis was conducted for the entire Walloon region and across five natural ecoregions. Main results show that forest cover – considered alone – explains around 30% of the biological water quality at the regional scale and from 15 to 70% across ecoregions. Furthermore, it is systematically positively correlated with higher biological water quality. When modulating this explanation power by spatial, local morphological variations, or population density, it is still above 10%. Partitioning variance shows that physico-chemical water quality is one of the main drivers of biological water quality and that anthropogenic pressures often explain an important part of biological water quality. The proportion of forest cover in each catchment at the regional scale and across every ecoregions except for the Loam region is more positively correlated with high water quality than similar analyses considering the proportion of forest cover in the riparian zones only. This suggests that catchment-wide impacts and a fortiori catchment-wide protection measures are the main drivers of river ecological water quality. Distinctive results from the agricultural and highly human impacted Loam region showed that remaining riparian forests have a positive impact on water quality and should therefore be preserved. However, as waterbodies are below 'good status' overall in this ecoregion, this is not sufficient to restore good ecological instream quality. On the other hand, this preponderance of a riparian forest cover link with biological water quality was not found in the other ecoregions and at the regional scale suggesting that protection measures at the catchment scale are required.

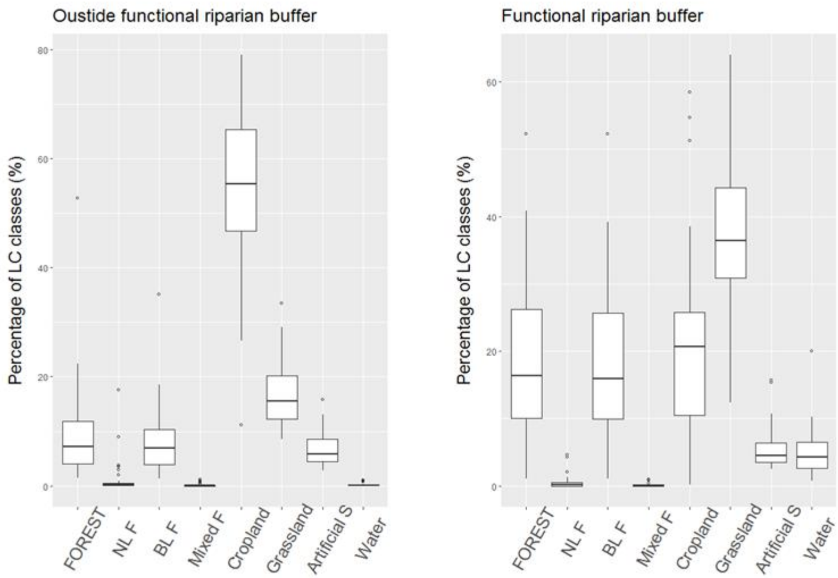
Supplementary Materials:

S1: LULC proportion outside and in the functional riparian buffer at the regional scale

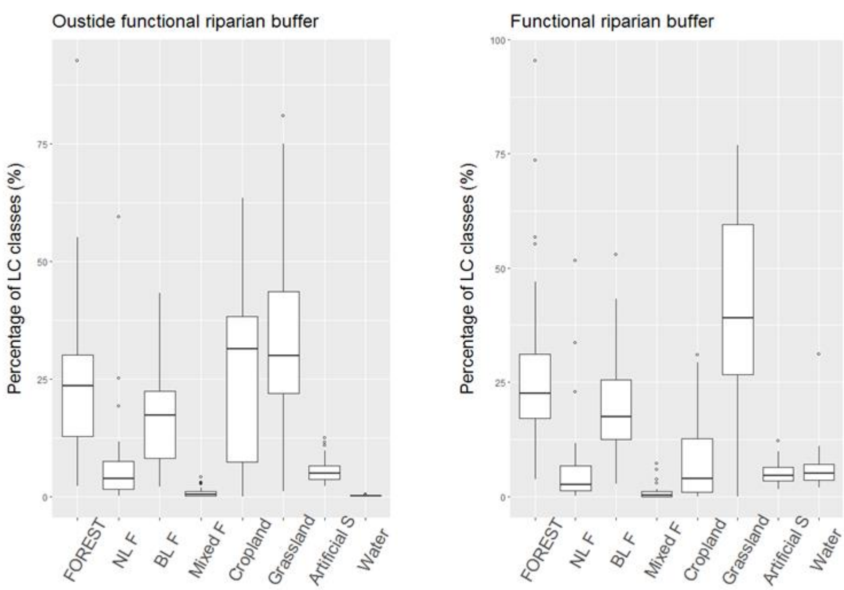
(A) Wallonia



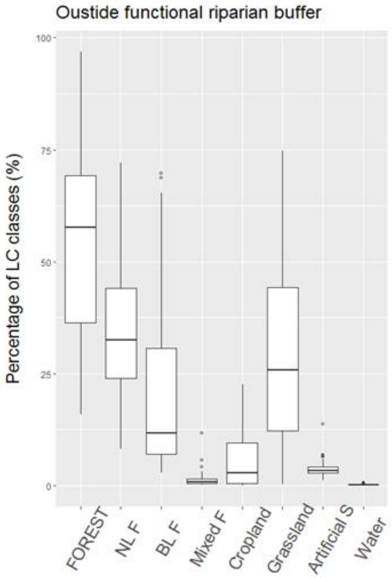
(B) Loam region

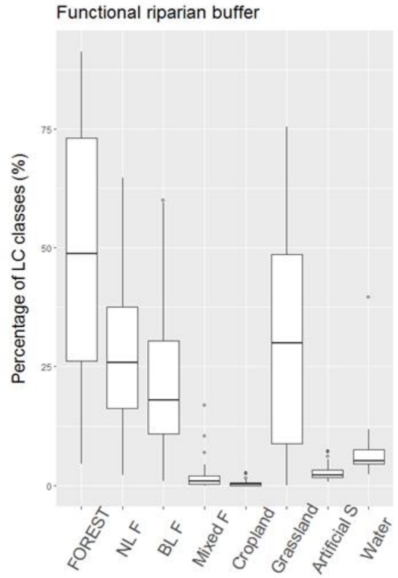


(C) Condroz

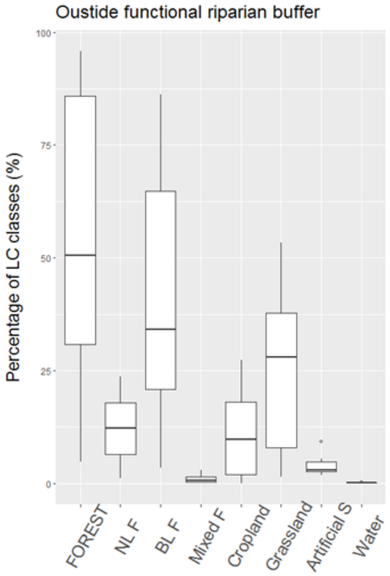


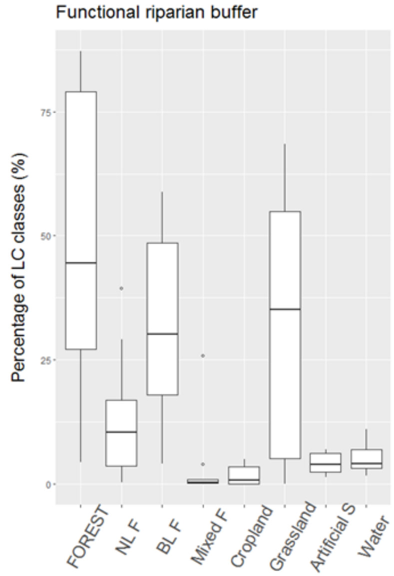
(D) Ardenne





(E) Belgian Lorraine





(E) Famenne

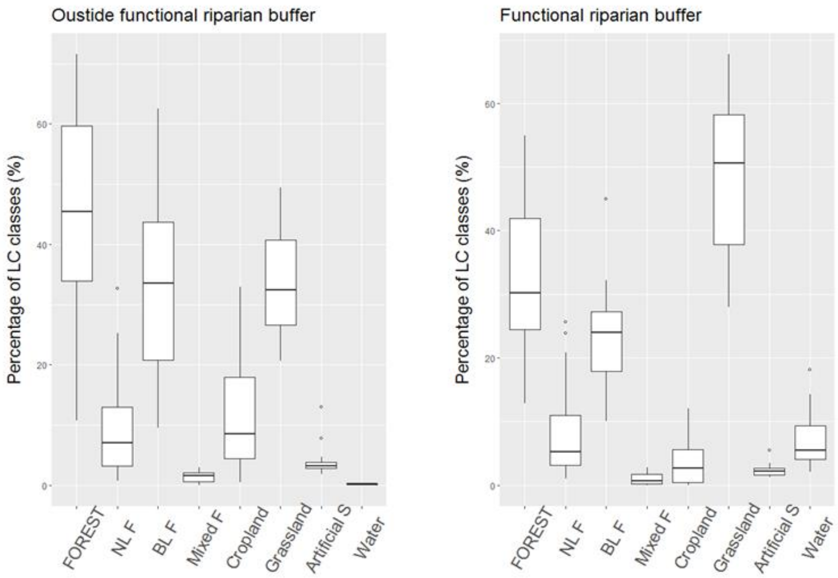
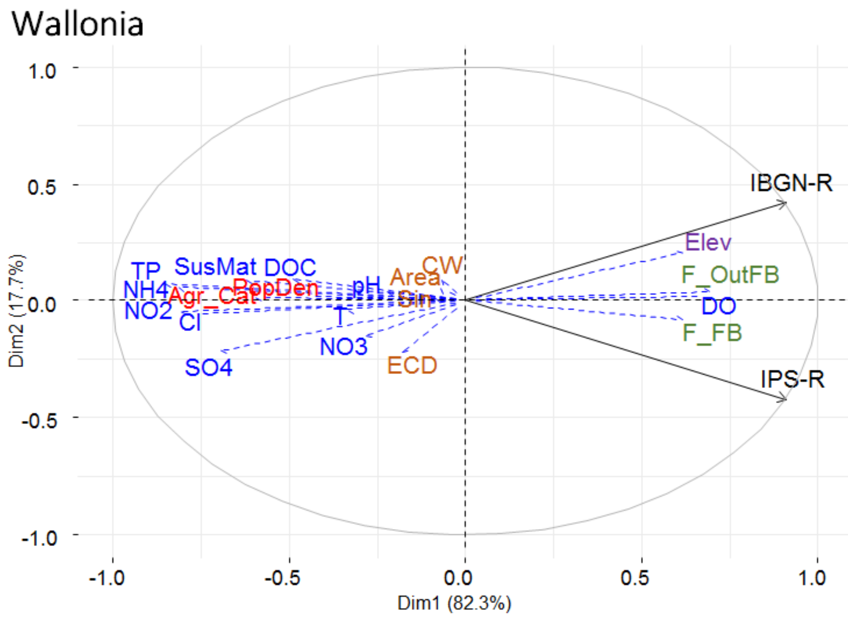
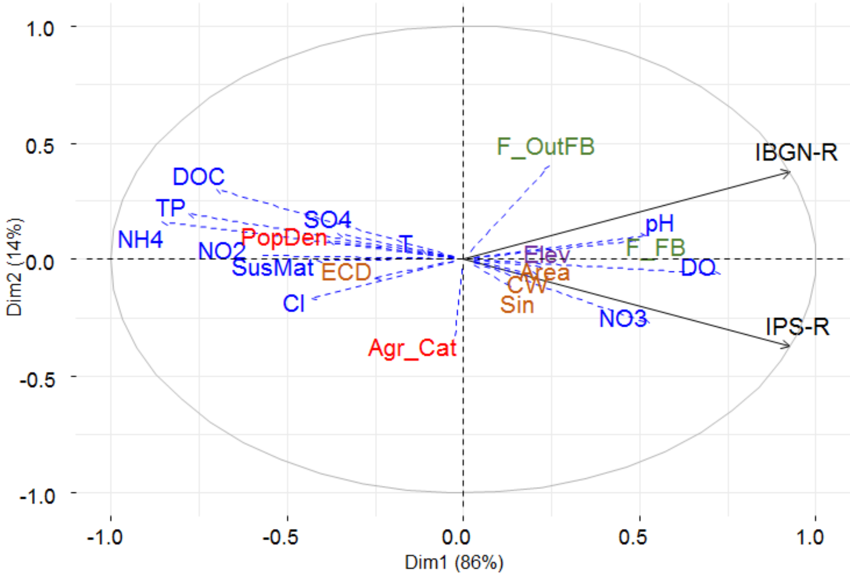


Figure S1: LULC proportion outside and in the functional riparian buffer at the regional scale (A) Wallonia and in the five ecoregions: (B) Loam region, (C) Condroz, (D) Ardenne, (E) Belgian Lorraine and (F) Famenne. With F : forest, NL needle-leaved , BL broad-leaved and S : surfaces.

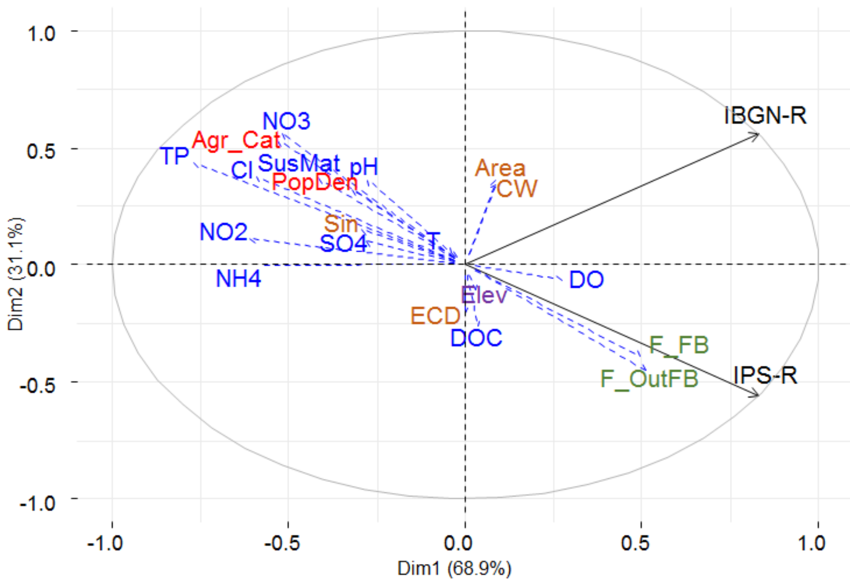
S2: Biological PCA biplot in Wallonia and the 5 ecoregions with physico-chemical variables, anthropogenic pressures variables, local morphology variables and elevation as supplementary variables



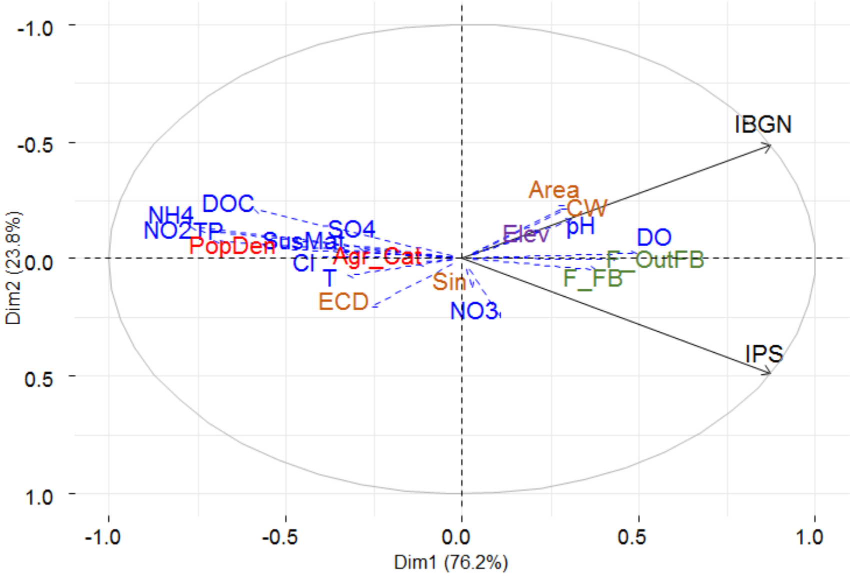
Loam region



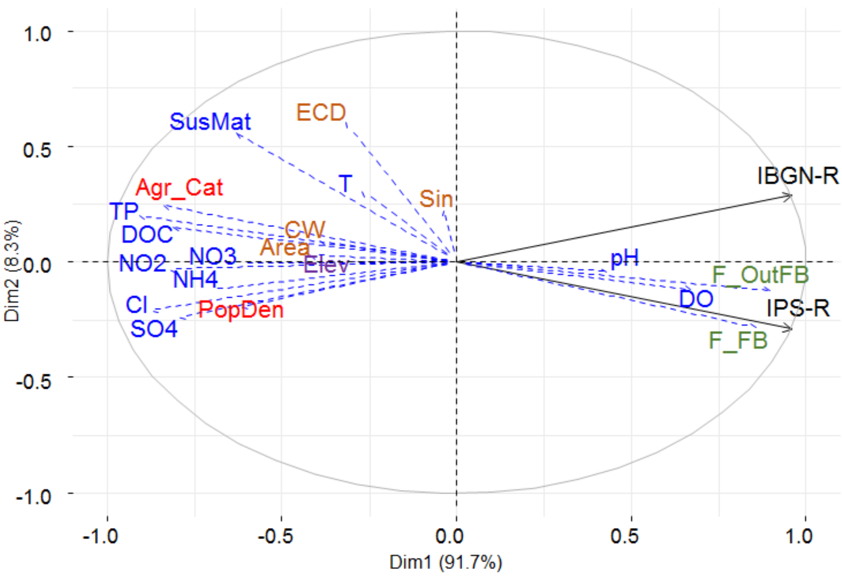
Ardenne



Condroz



Belgian Lorraine



Famenne

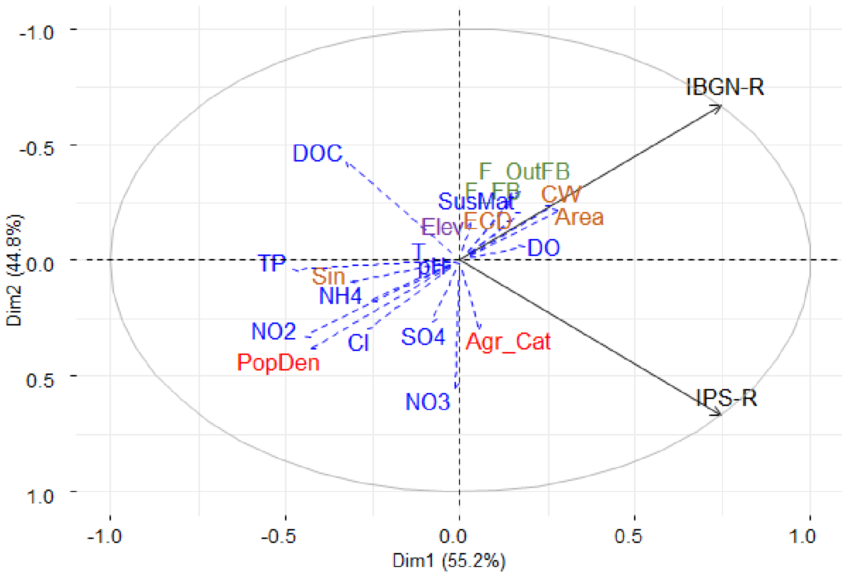


Figure S2: Biological PCA biplot in Wallonia and the 5 ecoregions with physico-chemical variables, anthropogenic pressures variables, local morphology variables and elevation as supplementary variables; with [Biological variables] IBGN macroinvertebrates index, IPS : diatoms index; [Physico-chemical variables] DO: Dissolved Oxygen, NO3: Nitrates, Cl: Chloride, SO4: Sulfates, TP: Total Phosphorus, NO2: Nitrites, NH4: Ammonium, DOC: Dissolved Organic Carbon, SusMat : Suspended Materials, [Anthropogenic Pressures] : Agr_Cat: proportion of agricultural land, PopDen: population density, [Local morphology variables] : Sin :sinuosity, ECD: emerged channel depth, Area : upstream catchment area, and Elev: elevation

Chapter 6 Discussion

This section is structured as follows: first, main results of the studies detailed in Chapters 3, 4 and 5 are synthetically presented and discussed. Second, contributions to the ES concept are presented. Then, insights for land management are suggested and finally, limitations of the study are discussed and perspectives suggested.

6.1 Preamble: on methodological objectives and forest cover in this study

As a reminder, transversal methodological objectives and work assumptions (see section 1.5) played a significant role in shaping this research. In particular, methodological objectives were: to produce easily replicable methods and to aim at broadening the scope of the research's findings in order to provide practical insight for land planning purposes.

In order to fulfil these methodological objectives along with the thematic ones, we chose to work at the catchment scale and study "**forest cover**" through a **proportion of forest cover in the upstream catchment**. This implies that we study various forests in terms of management, stand age, tree density, species combination, local conditions. Furthermore, studied catchments are what we called "real-life" catchments with mixed land covers – with high variability that we exploit through statistical analyses – and various local conditions that we discuss and attempt to control in the same analyses.

The first methodological objective was fulfilled through the development of methods based on, in principle, easily accessible public data monitored in many other European countries [whether regarding water quantity (discharges) or water quality (monitoring measures in the EU-WFD framework)]. Also, statistical methods were used to capture complexity in a robust and relatively simple way and to analyse large and highly variable sets of data. Second, to aim at a broad scope of the research's findings, we analysed "real-life" catchments (vs. small controlled pure LULC catchments) datasets with statistical methods. These catchments range from a few to hundreds km^2 and comprise mixed land covers. Choosing relatively large spatial extents of study (sub-regional to regional) and datasets (from 22 to 362 catchments) is a work assumption that better fits land planning processes that occur at similar scales, and allows deriving trends notably because of the variety of studied catchments (size, LULC combination, etc.). Finally, working at the ecoregional scale (Chapter 3), regional scale (Chapter 4 and 5) and distinguishing inter-ecoregion variability and common trends (Chapter 5) allowed to be more confident in the generalizability of some results. These characteristics of developed methods, mainly working at a complex landscape scale, sometimes render the interpretation more difficult. Study limitations and research perspectives are discussed further in this section.

6.2 Impact of forest cover on water related ES and attributes

6.2.1 Water quantity & timing

6.2.1.1 Instream water supply

The ES of instream water supply, studied in terms of quantity and timing (Chapter 3), was approached through three hydrological regime indicators: the specific volume, the baseflow index and the specific discharge exceeded 95% of time.

Multiple linear regression models taking into account rainfall, year effect, and land cover independent variables explain a significant part of the specific water volume (R^2 of around 0.70). According to our findings, **forest cover is negatively correlated with annual specific volume in our study area**. This negative link is in line with numerous studies that observed through paired-catchments designs a decrease in annual water yield when afforestation is operated or an increase in annual water yield when land covers such as grasslands are implemented in place of forests (Bosch and Hewlett, 1982; Brown et al., 2005). This can be explained by higher evapotranspiration rates of forest compared to lower vegetation as grass or arable land (Amatya et al., 2016; Granier, 2007; Office National des Forêts, 1999; Zhang et al., 2001). We believe this link is also partially explained by the location of the catchments. In our dataset, the catchments with low proportion of forest cover are located on higher zones being classified by Van der Perre et al. (2015) into the "Cold Ardenne" bioclimatic class (comprising the ''High Ardenne'' – Annual mean temperature of 7.7°C – and a part of the ''Mean Ardenne''), while catchments with high forest cover proportion are located into the "Warm Ardenne" (comprising the ''Low Ardenne'' and a part of the ''Mean Ardenne'', Annual mean temperature of 8.7°C). We therefore assume that temperature could also be part of and reinforce this effect of forest reducing more specific volume through potentially higher evapotranspiration rates than in lower vegetation.

We assessed the effect of forest type by testing the significance of a synthetic variable of LULC (second PC of LULC variables) on hydrological variables. Interpretation must again take into account that we compare mixed land covers with a more or less important gradient in forest cover (and other LULC) proportion. High values of this synthetic variable represent catchments with high needle-leaved forests proportion located on higher elevations and on soils with lower infiltration capacity and steeper slopes, whereas small values of this variable represent broad-leaved forests– in association with relatively high percentage of needle-leaved forests – and to a lesser degree, cropland on soils with better infiltration capacity (Figure 3-5). Infiltration capacity map takes into account soil texture, drainage characteristics, substratum and, when appropriate, percentage of stoniness (Demarcin et al., 2011). A significant positive link between this synthetic LULC variable and specific volume is demonstrated. We would expect the reverse, i.e. a higher annual and non-growing period water consumption from needle-leaved species, from the processes knowledge and experiments review in the literature (Nisbet, 2005). We interpret this by assuming that **local site conditions** (soil types, topography) **have a major impact on specific volume** (soils with low infiltration capacity and on steeper slopes being correlated with higher specific volume) **reinforced by the management option** (type of forests) especially when needle leaved forests on soils with bad natural drainage are artificially drained.

Despite the negative link between forest cover and the magnitude of streamflow, **this study shows a significant positive link between forest cover in low flows** and the specific discharge exceeded 95% of the time, annually and during the growing season. This could be a sign of a positive effect of forest on water supply in low flows conditions which could be explained by higher infiltration rates in forest soils (Calder, 2002). However, this is in opposition with the expected higher evapotranspiration rates in forests. This positive link between forest cover and the magnitude of streamflow in low flows is of extreme importance regarding riparian and aquatic habitat provision. This can also be directly linked to a positive impact on water quality as water dilutes nutrients and pollutants but also decreases stream temperature. **However, as this result is opposed to the general effect of forest on water yield** (i.e. specific volume indicator) and represents a real potential of positive effect of forest on instream water quantity in low flows, **we recommend further research** focusing of low flow periods to confirm this result.

Multiple linear models of baseflow index, which represents the way water infiltrates into the soil and returns to the stream, have low explanation potential. This testifies the **importance of other factors than land cover in explaining baseflow** such as highlighted by Price (2011). These factors are notably surface, subsurface topography and soil characteristics. Geology, one of the main factors influencing BFI, was, at least partially, controlled by the delineation of study area. Furthermore, in these models no significant link between forest and BFI is shown whether annually or during the growing period. Literature review does not provide us with strong assumptions of what we would expect in an "ideal" experimentation comparing numerous catchments while controlling other factors than land cover. Indeed some studies show a positive effect of forest on this indicator in accordance with the better infiltration capacity of forested soils (Bruijnzeel, 2004; Price et al., 2011), while others show the reverse effect linked to higher evapotranspiration rates (Hicks et al., 1991). Furthermore the differential impacts of forests compared to grasslands are less clear than with other land covers such as conventional crops regarding vegetation (Granier, 2007) and, obviously, artificial surfaces.

6.2.1.2 Water damage mitigation: flood protection

The ES of flood protection was approached through two hydrological indicators: the specific discharge exceeded 5% of time (Q_{05}s) and the flashiness index (FI). These hydrological variables are reversely linked to the flood protection ecosystem service. Regarding the specific Q_{05}, interestingly and unlike for low flow discharge as discussed above, we do not observe any positive significant link between forest and this indicator whether annually of during the growing period. Rather, we observe a slightly significant negative link during the non-growing period. The flashiness index which is the ratio of the 95th (i.e. $Q_{05}s$) by the 5th percentile (i.e. $Q_{95}s$) of specific discharge is negatively linked with forest cover which is a sign of the relative stability of the hydrological regime in forested catchments. This can be translated into a **positive link between forest and the flood protection ES**. However, further studies regarding the effect of forest cover on flood protection ES are necessary to deepen the knowledge initiated in the present study. Indeed, working at large spatial and temporal scales and especially on aggregated statistical discharges values rather than rainfall event is surely responsible for dampening effects. A better understanding of the underlying processes is also necessary to refine the interpretation.

6.2.1.3 Timing effect: Assessing the temporal distribution of flows

We tested the effect of timing (i.e. seasonal distribution of flows) of the instream water supply and flood protection by testing all models for annual values and seasonal values, i.e. April-September– for what we called the "growing period" – and October-March, the "non-growing period".

One main trend can be drawn: when variables are significant, developing a **seasonal model or the annual model does not change the direction of effect of the variable.** Variables linked to LULC were overall most significant in annual models than seasonal.

6.2.1.4 Main drivers of the ecosystem services of instream water supply and flood protection

Multiple linear regression allowed testing for the effect of forest on the ES of instream water supply and flood protection in terms of quantity and timing but also revealed the **importance of other environmental variables**. Overall, **rainfall** has a strong highly significant positive impact on water supply whether on total volume or in extreme conditions, and a logical strong negative impact on the BFI. **Rainfall remains as expected a main driver of the streamflow**. The "year" effect has a highly significant impact on each of the five hydrological indicators studied. We assume that this effect is a combination of several factors such as **climate variables** and, notably, temperature conditions (through its impact on potential evapotranspiration).

6.2.2 Water quality

6.2.2.1 Physico-chemical water quality

The study assessing the link between forest cover and physico-chemical water quality data at the regional scale allows drawing relatively clear conclusions (Chapter 4). The analysis of sub-catchment's LULC and the legal water quality status of streams shows that **sub-catchments with higher forest cover tend to achieve "good status" over the studied decade more often than sub-catchments with high cropland and/or grassland covers**. This is especially true for nitrogenous material and testifies that, despite the decrease in N input in agriculture since 1990, the EU-WFD target of "good status" is not yet fully reached. Sub-catchments with high grassland and cropland covers are also far more polluted with phosphorus than forested sub-catchments. Nevertheless, sites with good phosphorus status are more frequent than those with a good status in terms of nitrogenous materials.

Several insights can be derived from the quantitative assessment of the link between forest cover and water quality variables. First, **river size did not significantly affect the statistical relationship between LULC and water quality data**. This allows using data from the entire monitoring network, with a high diversity of catchment sizes and thus discharges. This also justifies the use of concentrations instead of loads, unlike some authors such as de Oliveira et al. (2016) suggest. This renders the methodology less complex and more easily replicable on such large numbers of sampling stations. **Seasonal and between-year effects** on the relationship between forest cover and physico-chemical water quality **were insignificant across the full decade**. This entails that the link between LULC and water quality reflects a background "multi-pollutants" load that can be considered as temporally stable. A potential seasonal effect as observed in other studies on particular relationships between particular variables and LULC (Álvarez-Cabria et al., 2016; Chen et al., 2016) might have been mitigated as the developed method treats several water quality variables together and as seasonal values are averaged values.

Analysis of the aggregated dataset over the studied decade showed that forest cover explains one third of the median water quality variability. Using elevation effect as a proxy for various environmental variables and as a mean for controlling spatial autocorrelation, we demonstrated an independent forest effect of 9.3%. Specifically, in this densely populated region with highly managed landscapes and forests, **sub-catchments with high forest cover** and with lower agriculture and grassland cover, **provide water with higher oxygen availability and lower concentrations of Ammonium, Nitrites, Nitrates, Total Phosphorus, Sulfates and Chloride and Dissolved Organic Carbon.** This is confirming previous findings and reinforcing papers stating that forest cover is associated with higher water quality (Fiquepron et al., 2013; Łowicki, 2012; Tong and Chen, 2002). This is probably partially due the "active" effect of forests that is underpinned by the protection against erosion resulting in water with less sediments and fewer nutrients and the nutrients filtration (Neary et al., 2009; TEEB, 2010). This is also due to the forest "passive" effect with lower anthropogenic pressure and nutrients loads on forest cover compared to agricultural and urban land uses.

Quantification of the partial effect of forest cover types (i.e., needle-leaved and broadleaved forests) on water quality and shared effects with other LULC and environmental variables represented by the elevation variable confirms a **clear effect**, independent from elevation, **of broad-leaved forest cover on water quality** (11%). The important effects of needle-leaved forest (29%) and cropland cover (39%) are largely shared with elevation and can therefore not be proven as independent effects. Indeed, a high proportion of water quality variability (21%) is shared between needle-leaved forest cover, cropland cover and elevation. Part of this variability is probably linked to the effect of forest cover but cannot be attributed to it with confidence.

6.2.2.2 Biological water quality

'Riparian forest' vs 'forest in the catchment' effect

In this study, we described biological water quality by two indices based on diatoms (IPS, "Specific Polluosensitivity Index") and macroinvertebrates (IBGN, "Standardized Global Biological Index") communities. Results show that **in Wallonia and for every ecoregion except for the Loam region, the forest cover proportion in the catchment** or in the area outside the functional riparian buffer **slightly better explains the biological water quality than the proportion of forest cover in the functional riparian buffer.** This trend is in line with several studies highlighting that catchment-wide disturbances are the most influential determinants of river ecological quality (Allan, 2004; Clapcott et al., 2012; Dahm et al., 2013; Marzin et al., 2012, 2013; Stephenson and Morin, 2009).

Forest cover effect on biological water quality

Forest cover is systematically related to higher values of both studied indices describing biological water quality (IPS-R and IBGN-R), corroborating studies associating forest cover with higher biological water quality (contrasting with agriculture and urban LULC) (Dahm et al., 2013; Ding et al., 2013; Kosuth et al., 2010; Theodoropoulos et al., 2015). **At the regional scale, forest cover explains a third of the variance in biological water quality (indicators),** and around 13% when controlling for spatial autocorrelation. This result is similar to the quantitative assessment of forest cover effect on physico-chemical water quality (see Chapter 4). Local morphology – approached through the sinuosity and the width of the local channel, emerged channel depth and upstream catchment area – impact on forest cover effect is small or even negligible at the regional scale and for the Ardenne and Condroz ecoregions. In the Belgian Lorraine or Loam ecoregions, the morphology effect is slightly more important. This can be explained by the contrasted morphological profiles of local rivers in the case of Belgian Lorraine. In the Loam region, comparing catchments' morphological characteristics with their proportion of riparian forest proportion reveals that catchments where some riparian forest remains have higher sinuosity and lower emerged channel depth. This trend has been highlighted by the characterization of riparian areas in Wallonia by Michez et al. (2017). Indeed, these authors demonstrate that forested riparian areas are associated with river reaches with lower channel depth and therefore higher ecological functionality. Population density effect on the relationships between forest cover and biological water quality is relatively important at every extent of study. Not surprisingly, population density is associated with lower water quality.

An important finding of the study, and more specifically of the quantification of the partial and shared effects between forest cover, anthropogenic pressures, and environmental variables, on biological water quality, is that **physico-chemical water quality acts as one of the main discriminating factor of biological water quality**. This result is in line with those of Dahm et al. (2013). This renders interpretation even more complex as Chapter 4 highlighted that physico-chemical water quality is influenced by LULC, and specifically forest cover. This is also in line with the nutrient enrichment shown in European studies and in Wallonia [e.g. European Environment Agency (2012) or SPW-DGO3-Direction de l'Etat Environnemental (2014)]. Forest cover explanation of water quality is also often shared with (i.e. inseparable from) anthropogenic pressures. This is especially true in Wallonia, and, at the ecoregion scale, for the Ardenne and the Belgian Lorraine. However, **the forest cover effect is in some cases interestingly relatively independent from anthropogenic pressures** such as in the Condroz and Loam region. In addition, complementary analysis in the highly anthropized Loam region (see details in sections 5.3.1 and 5.4.2) suggest a real independent effect of forest cover on biological water quality in these locations.

<u>*Water mitigation ecosystem service of 'protection against erosion'*</u>

Multivariate analysis shows that both biological water quality indices (i.e. IBGN-R and IPS-R) are systematically correlated to forest cover and dissolved oxygen and opposed to total phosphorus, ammonium, sulphates, nitrites, suspended materials, chloride, dissolved organic carbon, water temperature, and anthropogenic pressures (with a small difference for agricultural cover in the Fammene and the Loam region). The fact **that suspended materials are systematically negatively correlated to forest cover corroborates the positive effect of forest cover on the ES of protection against erosion** (Calder, 2002; Neary et al., 2009). We present this as a complementary result as we did not specifically quantified this effect.

6.2.3 Synthesis of thematic findings

Table 6-1. Synthesis of thematic findings (+: positive [effect], - : negative effect, 0 absence of effect, NS: non-significant, # non tested)

HES	***Instream Water supply***					***Flood protection (1)***	
Attribute	***Quantity***			***Quality***		***Quantity***	
				Physico-chemical	***Biological***		
Indicator	***Vs***	***$Q_{95}s$***	***BFI***	***9 variables***	***IBGN-R & IPS-R***	***$Q_{05}s$***	***FI***
Forest cover effect [reg. scale]	**-**	**+**	NS	34%; 9% when removing spatial autocorrelation	39%; 13% when removing spatial autocorrelation	NS	**-**
Forest type effect?	**+** NLF vs BLF	**+** NLF vs BLF	NS	Higher independent effect of BLF (11%) vs NLF (2)	#	NS	**-** from NLF vs BLF
Seasonality of effect?	Variation in models significance (3)			**0**	#	Variation in models significance (3)	
Riparian / catchment	#			#	Catchment scale (4)	#	
Main effect of forest cover on HES	**-** annually **+** in low flows			**+**		**+**	

(1) In this column, the sign of effects are linked to the quantity (i.e. opposed to the flood protection service) except for the line "Main effect of forest cover on HES" (2) NLF important effect is inseparable from Croplands & Elevation; (3) but not in effect direction, (4) except in the Loam region where riparian forest proportion is more linked to water quality than forest outside this buffer.

6.3 Contributions to the ES concept

6.3.1 Can our results be transposed to other ES classifications?

As mentioned above, the ES concept definitions and classifications are still evolving. When defining our objectives, we based our HES selection and terminology on Brauman et al. (2007) as other authors specifically studying forest hydrological ecosystem services did (Carvalho-Santos et al., 2014). Indeed, we assessed the instream water supply in terms of quantity, quality and timing and the water damage mitigation service of flood protection in terms of quantity and timing.

Initiatives of ES mapping and assessment are taking place at national or regional levels (see section 1.2.3) (Schröter et al., 2016; Stevens et al., 2015). In order to broaden the scope of the results and translate them into findings relevant to such frameworks, we established a correspondence (Table 6-2) between the studied ES categories and hydrological attributes and ES from the Common International Classification of Ecosystem Services (CICES), considered as the reference classification internationally (Haines-Young and Potschin, 2013). As we mentioned earlier, CICES groups ES hierarchically into sections, divisions, groups and, finally, classes. In the provisioning ES section of CICES, two ES have been studied here, surface water for drinking and non-drinking purposes. In the regulation and maintenance section of CICES, four ES have been studied related to the following "groups": mediation by ecosystems, liquid flows and water conditions.

Table 6-2. Hydrological ecosystem services studied in the present PhD thesis: correspondence between CICES (Haines-Young and Potschin, 2013; Turkelboom et al., 2013) and Brauman et al. (2007) based classifications

CICES for ecosystem accounting				***Brauman based classification***	
Section	**Division**	**Group**	**Class**	**ES category**	**Studied hydrological attributes**
Provisioning	Nutrition	Water	Surface water for drinking	Instream water supply	Quantity - Timing
	Materials	Water	Surface water for non-drinking purposes	Instream water supply	Quantity - Timing
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by ecosystems	Filtration/sequestration/storage/ accumulation by ecosystems	Instream water supply	Quality
	Mediation of flows	Liquid flows	Hydrological cycle and water flow maintenance	Instream water supply	Quantity - Timing
			Flood protection	Water damage mitigation	Quantity - Timing
		Water conditions	Chemical condition of freshwaters	Instream water supply	Quality

We matched the regulating HES classes “Filtration/ sequestration/ storage/ accumulation by ecosystems”, “Hydrological cycle and water flow maintenance” and “Chemical condition of freshwater” to the category of “instream water supply” of Brauman et al. (2007). We consider that these HES are directly linked to the dimension of quality of the instream water supply which Brauman et al. (2007) describe as provisioning services while specifying that ecosystems regulate water quantity and do not “create” water in a same way as e.g. ecosystems create timber. This discordance between classifications putting similar ES in different broad categories and in defining them with more or less details highlights again that classifications and terminologies can be variable across studies, evolving, and are in any case arbitrary. Another ongoing discussion concerns the sections (using CICES terminology, or broad categories in general) where water related services should appear, as “water” is an abiotic component of ecosystem (Haines-Young and Potschin, 2013). Should it be considered in another broad section covering abiotic services or be included in the three main CICES sections (i.e. provisioning, regulating and cultural ES)? Also, in the last CICES version, water appears under provision and regulation services whereas a discussion about the role of ecosystem linked to water could lead to grouping water related ES only in the regulation section as living processes clearly play a role in regulating its quantity and quality and not so much the water “production”. As scientific users of the ES concept, and in line with Haines-Young R. and Potschin M. (2016) view, we believe that this plurality of definitions and classifications does not represent any problem when studying particular ES in particular geographical conditions because the process of defining ES brings in itself interesting reflexions. In our study, the objective of describing the forest cover effect on water related ecosystem services in order to derive relevant information for land planning and forest cover preservation in particular, is not affected by this plurality of classifications and definitions. However, choosing one classification over another will have an impact on the broader ES assessments that will be derived from it. When considering frameworks of national accounting and mapping (European Commission et al., 2014; Maes et al., 2013), it is important to be consistent about choices made to enable comparison between geographic zones and studies repeated over various time steps. Therefore, we recommend further research and harmonization in ES classification and terminology in that context.

6.3.2 Land cover proxies for ES assessment?

Regarding the debate of using a LULC-based matrix model approach – versus more advanced methodologies such as detailed and precise simulation and process models or direct mapping (based on survey and census providing spatially detailed ES distribution) – to derive indicators used to map water related ecosystem services, our findings vary according to the hydrological attributes and HES studied. Indeed, with regard to HES of instream water supply or flood protection in terms of quantity and timing, results show that other factors than land cover strongly impact water flows at the catchment scale. In particular, analysis of the effect of the type of forests (NL vs BL) on specific volume suggests an effect of terrain topography but also soil types and, we assume, related forest management options (artificial drainage but also forestry in general). Moreover, low R^2 of the baseflow index models show that there are other, non-quantified here, important factors acting on this aspect of water supply. We touch here one limitation of working with "real-life" catchments with mixed LULC, as even if forests are the dominant land cover, their effects on hydrological indicators may be dampened by the effects of other land covers. Another factor is the type of forest, in particular the differences between needle-leaved and broad-leaved species and their respective water use strategies, which induces seasonal differences on catchment water balance (the partitioning of actual evapotranspiration fluxes in particular). Last but not least, the effect of 'year' (i.e. climatic characteristics) and rainfall are highly significant in most models showing the importance of climatic conditions (rainfall and potential evapotranspiration and associated climatic variables) on HES studied. In the current context of climate change, inducing more frequent spring and/or summer droughts and warmer winters, this draws attention to the adverse impacts it may generate towards water related ES. This suggests that, unless regarding "pure" forested catchments such as e.g. in Canada, using spatially unvarying land cover proxies as indicators of the ES of instream water supply is, at best, a very crude estimate of the underlying processes and, at worse, giving a false representation of such processes (by giving indicators of wrong sign, and/or that cannot be compared to other indicators). This is especially worrying if the mapped extent using such method is large. In this context, we recommend further research integrating at best local condition factors (soil characteristics, slopes, climate, etc.) where each land cover is actually located (and not in the catchment overall) in order to come up with integrative proxies indicators of ecosystem services. Climatic variables and climate change should also be taken into account given their high influence on these HES.

Regarding the HES of instream water supply in terms of quality, our results allow us to be more confident in land cover proxy mapping when associating forest cover with a global highly positive influence on this service. However, this should be nuanced by the fact that this effect is opposed to and largely inseparable from a negative impact of pressure LULC, i.e. agricultural and urban LULC, even if results in the Loam region encourage us to mention a real independent forest effect. Similar studies should be replicated in time so the evolution and/or stability of the effect could be assessed in line with the evolution of management practices in agricultural land.

6.4 Insights for land management

Though we recommend further research concerning the effect of forest cover and other factors (topography, soil types) on water supply in low flow and on flood protection for various rainfall events, we can still derive some interesting insights from this study. For example, the inter-annual variability (representing climatic characteristics) and rainfall were highly significant in most of the models showing the importance of climatic condition on instream water supply and flood protection ES. In the current context of climate change, inducing more frequent spring and/or summer droughts, this draws attention to the adverse impacts it may generate towards water related ES. In Europe, this is especially true at the extreme of gradients with, at the north-western “wet-end”, even more precipitation and, at the south-eastern “dry end” less precipitation under simultaneously increased temperatures (Bredemeier, 2011; IPCC, 2007). In Belgium, located at the centre of this gradient, the situation could seem intermediate but some climate model simulations show a clear shift in the precipitation pattern with an increase during winter and a decrease during summer (Baguis et al., 2010). This will impact instream water supply but also groundwater supply (Woldeamlak et al., 2007). Land management should consider study of ecosystems and LULC impact on water related ES in the context of climate change to favour instream water supply while protecting against erosion and flood. In addition to this, foresters will have to face climate change effects in the forest itself and the forest sector. As synthetized by Sousa-Silva et al. (2016), these will be: (i) an increase in the frequency and intensity of tree diseases and pest outbreaks (Dale et al., 2001), (ii) a modification of the potential distribution ranges of tree species (Bell and Collins, 2008); and (iii) warmer growing seasons and rising CO_2 concentrations, which, in the short term, will enhance forest production where soil nutrient and water availability allow. However according to Campioli et al. (2012), this will likely not occur in the Belgian forest regions of Ardennes and Campine which are implemented under nutrient-poor or under water-deficient conditions. In order to face these projections and especially the difficulties in keeping a forest healthy, we recommend, along with the research on adaptability of tree species/soil conditions under climate change projections, further research on water supply ES (whether instream or in the subsoil) from both the demand and the supply points of view to better anticipate and face these changes.

Regarding water quality, analysis of forest cover (vs other LULC) impact on biological water quality at different extents and considering different forest spatial units (i.e. riparian or at catchment scale) brings interesting insights. Indeed, forest cover impact is in some cases proven to be relatively independent from anthropogenic pressures such as in the Condroz and Loam region. In addition, complementary analysis in the Loam region suggests a real independent effect of forest cover on biological water quality in these locations. Indeed, the fact that forests are mostly present in the functional riparian buffer in this ecoregion while relatively absent in the rest of the catchments combined with agricultural models being non-significant at the catchment scale let us believe that the computed forest link with biological water quality represents a "real" forest effect. Furthermore, study of the functional riparian buffer LULC explanation power of water quality revealed that only cropland cover model was significant (i.e. not grassland cover) and explained a twice-lower proportion of variability than forest cover. Consequently, in these catchments and in similar catchments outside these regions, riparian forests should be protected because of their positive effect on biological water quality. This is in line with Tran et al. (2010) findings showing a stronger correlation between LULC and stream water quality at the 200-m riparian buffer than that of the watershed. The underlying processes they assume as responsible for this is that since surface water contamination is highly dependent on storm water runoff, contaminants located in close proximity are more likely to reach water bodies than those located at a further distance. They also suggest that the presence of a riparian buffer zone between streams and agricultural and urban areas might reduce contamination from non-point source pollution. On the other hand, the fact that we did not detect this preponderance of a riparian effect in the other ecoregions and at the Walloon scale suggests, as noted by Stephenson and Morin (2009) or Harding et al. (1998), that maintenance or preservation of habitat fragments in the riparian zone will not be sufficient to preserve ecological instream quality from catchment-wide impacts. Rather, this requires protection measures at the catchment scale in addition to riparian ones such as increasing forest cover in problematic zones and decreasing pressures by adopting more environmentally responsible agricultural practices. Further studies are necessary to come up with practical recommendations notably linked to the position of forest in the catchment and management practices (whether in forest or agricultural LULC). Example of protection or restoration measures suggested by our results can be found in Natural Water-Retention Measures (NWRMs), emphasized by the EU Biodiversity Strategy to 2020 (European Commission, 2011) and promoted by the EU Forest Strategy (Zal et al., 2015).

These are defined as "measures to protect and manage water resources and to address water-related challenges by restoring or maintaining ecosystems, natural features and characteristics of water bodies using natural means and processes" (European Commission, 2013; Zal et al., 2015). Among the 53 different NWRMs suggested by the Commission's study on Natural Water Retention Measures, 14 are forest-related but obviously, others are linked to agricultural practices.

6.5 Limitations of the study and perspectives

6.5.1 Impacts of methodological choices

Some limitations of the study can be pointed out notably from the methodological choices we made. Working with "real-life" catchments (vs classical experimental pair-wised approaches working with smaller catchments where factors external to the object of study are attempted to be controlled) complicates the drawing of clear lessons from this study. For example, other factors correlated with forest cover impact on hydrological ecosystem services in terms of quantity and timing (e.g. slope, soil infiltration capacity, tree species use of water, phenology…). On the other hand, "ideal" experimental studies that could study each factor separately are hardly possible in Wallonia given the heterogeneity of landscapes, or are at really small-size catchments scale or stands. Taking such an approach of studying stands or small-size catchments would then lead to an upscaling problem. One could think about an intermediate methodology that would, within similar public datasets, select some catchments based on different criteria related to factors highlighted in the present study: slope, phenology and infiltration capacity. This could complement and deepen the interpretation of the present study. In that case, the fact that some of these factors are inseparable from each other and from certain LULC classes should be kept in mind.

Still with regard to Chapter 3, the selected indicators of hydrological flows are statistics characterizing the overall hydrograph. Further research could concentrate on specific rainfall events and further detail the behaviour of the catchment to provide insight of the effect of forest cover on HES at the event scale. This is particularly true for the service of flood protection on which the effect of forest cover should differ according to the intensity of each rainfall event (Lana-Renault et al., 2011).

The methodological design and data used to capture the forest cover effect on physico-chemical and biological water quality does not allow for isolating quantitatively a potential "active" effect of the forest (i.e., water purification *per se*) from the "passive" one being directly linked to the degree of pressure of each LULC on these ES. Multivariate analyses developed in Chapter 2 and Chapter 3 present the advantage of capturing the relationships between several water quality variables and forest cover. However, this implies that conclusions remain rather general. This arguably represents an advantage for land planners who are dealing with multi-pollution sources and complexity of ecosystem processes, but does not allow us to discuss one particular variable in detail. Furthermore, for studies aiming to clearly focus on seasonal effects of forest cover on water quality, we recommend specific and regular spatio-temporal sampling while focusing on homogeneous groups of pollutants regarding their seasonal variability [see e.g., Johnson et al. (1997)].

We believe that the developed methods allowed for studying ecosystems functions and HES at a regional scale and provide land planners with insights potentially contributing to a more sustainable resource management. In response to the above-raised limitations and to deepen the results' interpretation, especially with regard to processes underpinning HES supply, we suggest complementing the present study with others shaped differently, whether in terms of object of study, sampling scheme, spatial extent, type of collected data, type of methods, etc.

6.5.2 A small contribution to understanding the ES cascade

This study has been undertaken in the frame of a 'stand-alone' PhD and choices had to be made to limit its scope. Still, it captures part of the complexity concerning the whole 'supply' side of the ES cascade model (see Figure 1-4) in the case of HES provided by forest cover. This could be complemented by other studies, as suggested in the previous sections. However, if we step back and enlarge the scope by considering the broad ES concept along with its purpose of contributing to a better resource management for increased human well-being, contextualization elements must be provided.

HES trade-offs and synergies

Only few HES have been considered in this study. We can only hint at one trade off when considering water supply and water mitigation ES of flood protection. Indeed, if we desire a bigger amount of water available to extraction and as much as possible of water in low flows, we do not desire too high quantity of water flows in relation to flood and erosion protection. When studying ES bundles, which is highly recommendable in land planning support in order to avoid unwillingly managing for one ES at the expense of others, one must consider HES at adapted spatial and temporal scale based on reliable biophysical assessment but also with a focus on matching supply and demand.

ES supply but what about demand?

In the present study, we brought insights about the potential of forest in delivering HES, working on the 'supply' side of the cascade (Figure 1-4). However, studying the demand side of these services and see how they spatially match would be interesting, especially e.g. with regard to the services of flood protection and erosion protection as they are of concern in regions such as the Loam region. Furthermore, these services have been ranked as of prime importance to stakeholders of this ecoregion (Baptist et al., 2016; Fontaine et al., 2013). This reinforces the interests to match the demand and supply sides and thus the interest to have biophysical assessments (supply side) as accurately as possible.

Biophysical assessment? Only a part of "integrated valuation"

We assessed the impact of forest cover on some particular HES from the 'supply' point of view and through a biophysical assessment. This represents only a part of the big picture and we recommend broader studies integrating different kind of valuations for best use of the ES concept in land planning [see Jacobs et al. (2016) contribution regarding a new valuation school in ES assessment]. Integrated valuation explicitly aims at including the multiple values (ecological, socio-cultural and economic values) and worldviews in a coherent and operational framework pursuing a societal rather than (only) academic impact (Gomez-Baggethun et al., 2016).

Chapter 7 Conclusions

The main objective addressed in this research is the study of the impact of forest cover on hydrological ecosystem services in Wallonia (Belgium). In particular, the effect of forest cover on instream water supply and flood protection was studied in terms of quantity, quality and timing. Datasets, methods and spatial scale of study were chosen to provide insights for water and forest management policies.

Main results show (Chapter 1) an assumed negative effect of forest cover on instream water supply service in terms of quantity when studying water yield whereas a significant positive link between forest cover and streamflow was demonstrated in low flows. Studying baseflow and forest type effect let us assume that local site conditions (soil types, topography and forest management) have a major impact on water supply. Regarding flood protection, forest cover showed a negative impact on the flashy behaviour of the catchment that could be linked to a positive effect on the flood protection ES. Climatic factors and in particular rainfall are often significant and can be considered as main drivers of these ES. Regarding the instream water supply in terms of quality, several insights were drawn from this study. First, the study assessing the link between forest cover and physico-chemical water in Wallonia (Chapter 4) showed that forest cover explains about one third of the variability of water quality (9% when spatial autocorrelation is controlled) and is positively correlated with higher quality water. Results also show that unlike needle-leaved forest cover, broad-leaved forest cover presents an independent effect from ecological variables and explains independently 4.8% of water quality variability while it shares 5.8% with cropland cover. Studying effect of forest cover on biological water quality (through diatoms and macroinvertebrates indices, Chapter 5) showed that forest cover – considered alone – explains around one third of the biological water quality at the regional scale and from 15 to 70% depending on the studied ecoregion. Forest cover is systematically positively correlated with higher biological water quality. Partitioning variance shows that physico-chemical water quality is one of the main drivers of biological water quality and that anthropogenic pressures often explain a relatively important part of biological water quality. The proportion of forest cover in each catchment at the regional scale and across every ecoregions except for the Loam region was more positively correlated with high water quality than when considering the proportion of forest cover in the riparian zones only. However, distinctive results from the agricultural and highly human impacted Loam region show that riparian forest have a positive impact on water quality and should therefore be preserved where they are left (i.e. in riparian zone).

These results allowed us to come up with recommendations regarding ES assessment and mapping on one hand and land planning on the other hand. Results regarding forest cover effect on studied HES in terms of quantity and timing made us question the use of land cover proxies to assess and map hydrological ES at a complex landscape scale. However, the strong link between forest cover in catchment and water quality allows to be more confident when using simple land cover proxies to map services related to water quality.

Regarding land planning recommendations, results at the regional scale and across ecoregions lead to recommend riparian forests protection in the Loam region (where they are left) but as the overall forest catchment effect on water quality (whether physico-chemical or biological) suggests that catchment-wide impacts and *a fortiori* catchment-wide protection measures are the main drivers of river ecological water quality. Forest should therefore be preserved (and/or restored) at least in riparian zones of highly intensive agricultural areas such as the Loam region, knowing that this will not be sufficient to bring water bodies to "good quality status" referring to EU Water Framework Directive standards. Also, alternative agricultural practices should be adapted in order to decrease pressures on water.

Given our results, we recommend further research, notably focusing on forest cover impact on instream water supply service in low flow periods to confirm or infirm the positive effect found in this study. We also strongly recommend further studies on flood protection to refine interpretation of processes underpinning ES supply. These studies could focus on extreme rainfall events rather than statistical values. Given its replicability, this method could be applied in other regions were similar data are available as in other European countries. We also recommend further research using the ES framework in order to enlarge the scope of study to the demand of ES or to the study of potential trade-off and synergies between hydrological services and provision or cultural services.

We believe that this study, anchored in the ES approach, contributes to answer questions related to Ecohydrology at a scale meaningful for land planning processes. While acknowledging the complexity and difficulty to study ecosystem functioning at the landscape scale, we are truly convinced that these approaches at large scale, corresponding to policy planning, should be carried out to complement finer scale studies. Both types of approaches are necessary and should be integrated in order to capture the underlying complexity of ES provision and contribute to a better resource management.

Chapter 8 References

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